

**EMPIRICAL
NUTRIENT -- PHYTOPLANKTON
RELATIONSHIPS**

January , 1978



Ontario

**Ministry
of the
Environment**

The Honourable
George R. McCague,
Minister

K.H. Sharpe,
Deputy Minister

Copyright Provisions and Restrictions on Copying:

This Ontario Ministry of the Environment work is protected by Crown copyright (unless otherwise indicated), which is held by the Queen's Printer for Ontario. It may be reproduced for non-commercial purposes if credit is given and Crown copyright is acknowledged.

It may not be reproduced, in all or in part, for any commercial purpose except under a licence from the Queen's Printer for Ontario.

For information on reproducing Government of Ontario works, please contact ServiceOntario Publications at copyright@ontario.ca

EMPIRICAL NUTRIENT-PHYTOPLANKTON

RELATIONSHIPS

P.J. Dillon, K.H. Nicholls and W.J. Kennedy

Limnology and Toxicity Section
Ontario Ministry of the Environment
Box 213, Rexdale, Ontario

January, 1978

PREFACE

The contents of this report formed the basis for a lecture given by one of the authors at the Freshwater Institute in March, 1977. The work described summarizes some important aspects of the research carried out by the Limnology and Plankton Taxonomy Units since their inception; thus it was decided that it would be useful to release this information in a report form.

MOE
EMP
APET

TABLE OF CONTENTS

	Page
PREFACE.....	ii
ABSTRACT.....	iv
DISCUSSION AND SUMMARY.....	1
REFERENCES.....	70
APPENDIX A.....	73

ABSTRACT

Freshwater management has suffered from a lack of simple predictive models. It is desirable that such models be easily manipulated by the lake manager and offer him credibility and confidence when it comes to making decisions. Improved modelling methodology for phosphorus is presented along with further evidence to support the hypothesis that phosphorus is the key element in the biogeochemistry of north temperate lakes. It is stressed that the lake and its watershed are the unit under consideration when employing such models. Once phosphorus concentration is determined, other water quality parameters such as algal biomass and hypolimnetic oxygen depletion rate can be predicted. Ultimately, one would like to employ a master variable such as phosphorus concentration from which to extrapolate many other aquatic ecosystem parameters but the complexity of dynamic natural systems limits this goal. Further discussion emphasizes the complexity of nutrient-phytoplankton relationships in lakes and that, as a result, lake management techniques should still be practiced with caution.

DISCUSSION AND SUMMARY

One of the important questions that our group in the Ministry of the Environment is trying to answer is "How much development, in terms of houses, cottages, condominiums or campgrounds etc., can a given body of water withstand before degradation of water quality occurs beyond an acceptable level or, conversely, how much will any specified human activity quantitatively effect the water quality of any body of water associated with it?", and to describe our present attempts to answer this question, we have to review a number of points.

It is now widely recognized that P is the most important element governing the trophic status of most lakes. We have results from three studies which reinforce this hypothesis very clearly (Nicholls et al. 1977, Dillon et al. 1978, Scheider and Dillon 1976).

The first example is Lake Erie's Western Basin. The accelerated eutrophication of Lake Erie over the period 1919 to 1965 has been well documented by Davis (1964), Verduin (1964) and Hohn (1969) who summarized long-term phytoplankton records for the lake. By the late 1960's and early 1970's, attempts were already underway to halt the eutrophication of Lake Erie by decreasing the phosphorus load to the lake. It had been estimated that phosphorus of detergent origin contributed about 40% of the total phosphorus loading to Lake Erie, prior to legislative action by both Canada and the U.S.A. between 1971 and 1973 to limit the phosphorus content of detergents to 8.7% (as P, by weight). At weekly intervals since 1967, phytoplankton analyses have been performed on water collected at the Union Water Treatment Plant in the Western Basin of Lake Erie. Since 1971, all classes of algae have declined sharply and have accompanied decreased concentrations of total phosphorus in the Western Basin near-shore area (samples collected also by the Union Water Treatment Plant) and decreased phosphorus loading from the Detroit River (Michigan Department of Natural Resources data).

Since the 1967-1970 period, when average phytoplankton density was 5010 A.S.U. ml^{-1} phytoplankton density had fallen about 42% to 2900 A.S.U. ml^{-1} by 1975. There was no further change in 1976 and by 1977 algal density had increased slightly (Fig. 1). Decreased densities of diatoms account

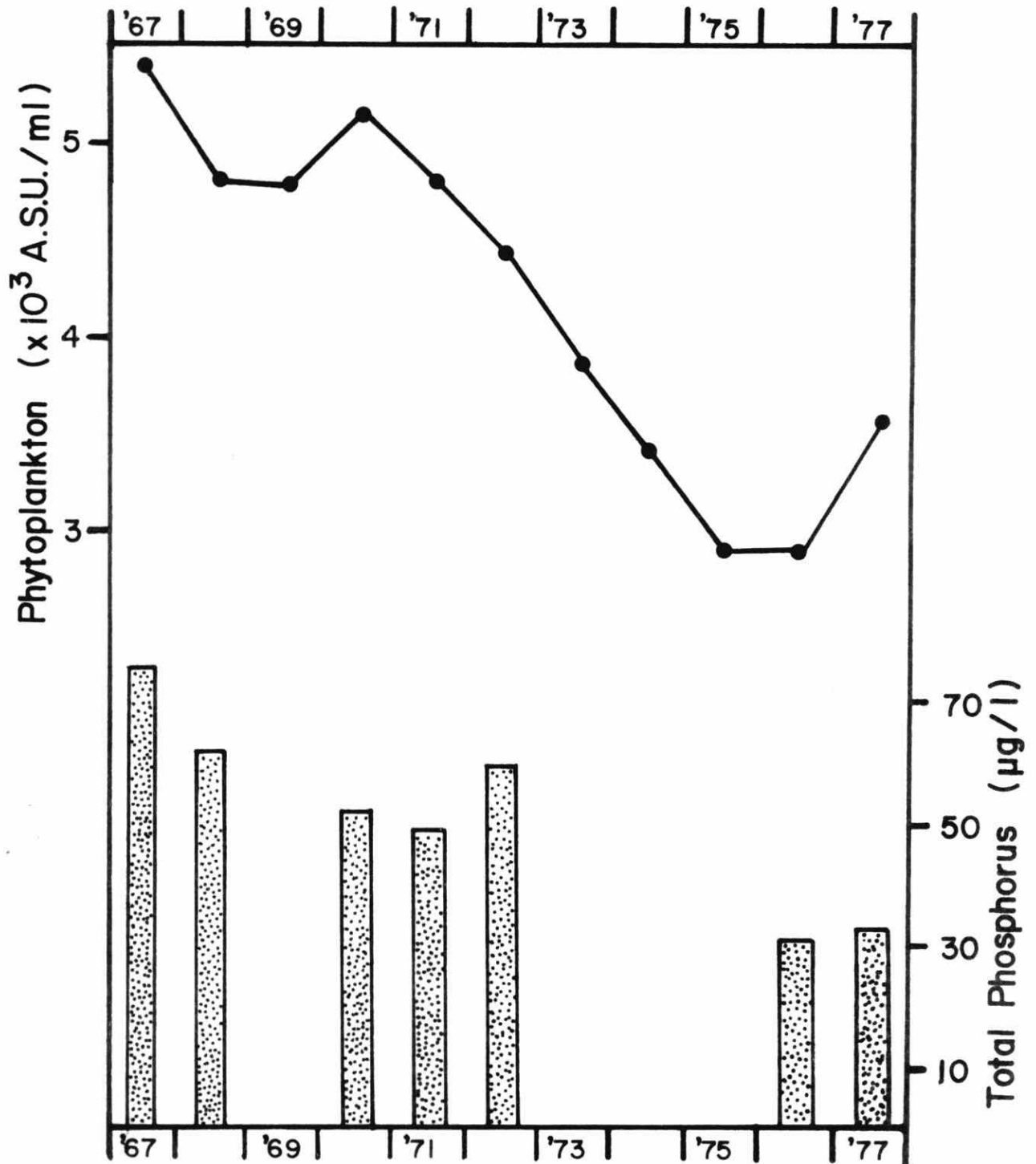


Fig. 1. Average annual phytoplankton density (Areal Standard Units/ml) and total phosphorus concentration in near-shore western Lake Erie, determined from samples collected approximately weekly at the Union Water Treatment Plant at Kingsville, Ontario (updated from Nicholls *et al.* 1977).

for most of the decrease in total phytoplankton, since this group comprised 65-80% of the total during the long-term study. These findings should offer encouragement for the "reclamation" of other highly eutrophic inshore waters of the Great Lakes such as Lake Ontario's Bay of Quinte, Lake Huron's Saginaw Bay and Lake Michigan's Green Bay, given similar controls on phosphorus loading.

The second example is Gravenhurst Bay (Fig. 2) which lies on the Precambrian Shield and which has been extensively studied from 1969-77 (Michalski and Nicholls 1975; Michalski et al. 1975, Dillon et al. 1978). In late 1971, phosphorus removal was initiated at the town's sewage treatment plant as a test case for nutrient diversion effects upon a lake. From Table 1 it can be seen that by 1975 removal was effectual in reducing P concentration in the Bay to 45% of the pre-removal average, chlorophyll a concentration to 51%, the quantity of phytoplankton to 48% and the areal hypolimnetic oxygen demand to 50%. From Fig. 3 it is clear that the areal hypolimnetic oxygen demand in Gravenhurst Bay was much greater in 1969 than 1975.

By 1975, the Bay had shown good overall improvement, however, in June of 1976, a massive bloom of Aphanizomenon occurred. Apparently this was the result of the failure of P removal at a second sewage treatment plant which was brought into operation in October 1975 to accommodate the expansion of Gravenhurst. About 50% of the town's wastes were directed through the new sewage treatment plant which was only operating at 20-30% P removal efficiency instead of the expected 80-90%. The failure continued from October, 1975 to April, 1976 at which time it was rectified. The results were severe (Table 2): chlorophyll a levels and areal hypolimnetic oxygen demand were as high and phytoplankton quantities higher than during the pre-removal period. P concentrations and Secchi disc values were only slightly worse than in 1975 but these mean values do not adequately represent the radical changes within the Bay. For instance, even though Secchi disc was the lowest ever recorded in the early summer of 1976 (Fig. 4), by fall it was the best. The same pattern showed up in chlorophyll a levels (Fig. 5) and P concentrations (Fig. 6). Similarly, the phytoplankton community

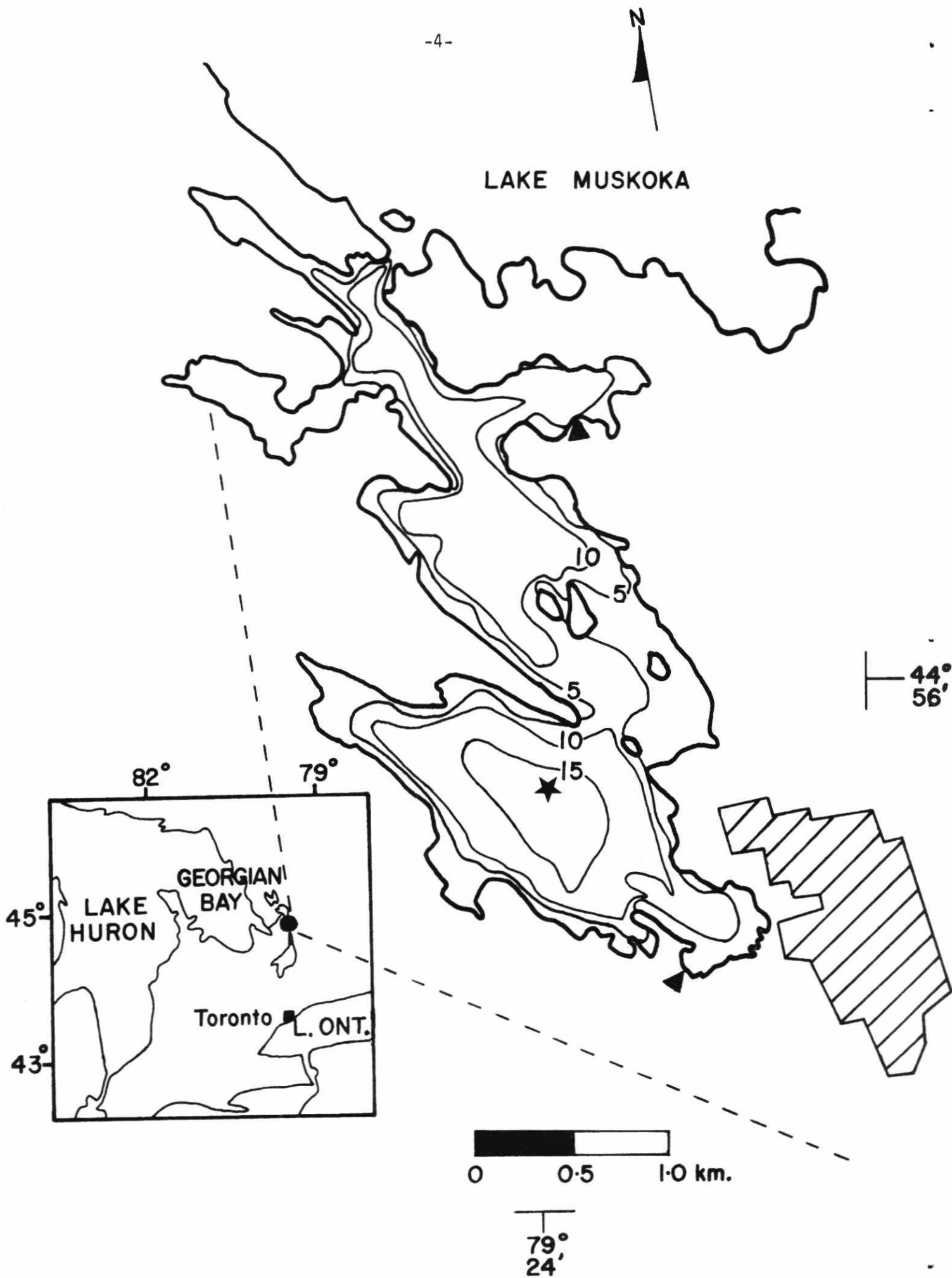


Fig. 2 Location and morphometric map of Gravenhurst Bay. Sewage treatment plants are marked with a triangle, sampling station with a star.

Table 1: Mean total phosphorus, chlorophyll a, Secchi disc depth (S.D.), quantity of phytoplankton and areal hypolimnetic oxygen deficit (AHOD) in Gravenhurst Bay for three years prior and four years following P removal (from Dillon et al. 1978).

Year	[P] (mg.m ⁻³)	[chl <u>a</u>] (mg.m ⁻³)	S.D. (m)	Phyto. (ASU.ml ⁻¹)	AHOD (mg.cm ⁻² .d ⁻¹)
1969	42	10.6	2.6	1780	0.053
1970	39	5.1	3.1	1620	0.047
1971	<u>52</u>	<u>13.8</u>	<u>1.9</u>	<u>3620</u>	<u>-</u>
Mean	44	9.8	2.5	2340	0.050
1972	35	8.1	3.1	1410	-
1973	33	6.9	3.2	1470	0.032
1974	25	5.0	2.7	1710	0.043
1975	20	5.0	3.9	1120	0.025

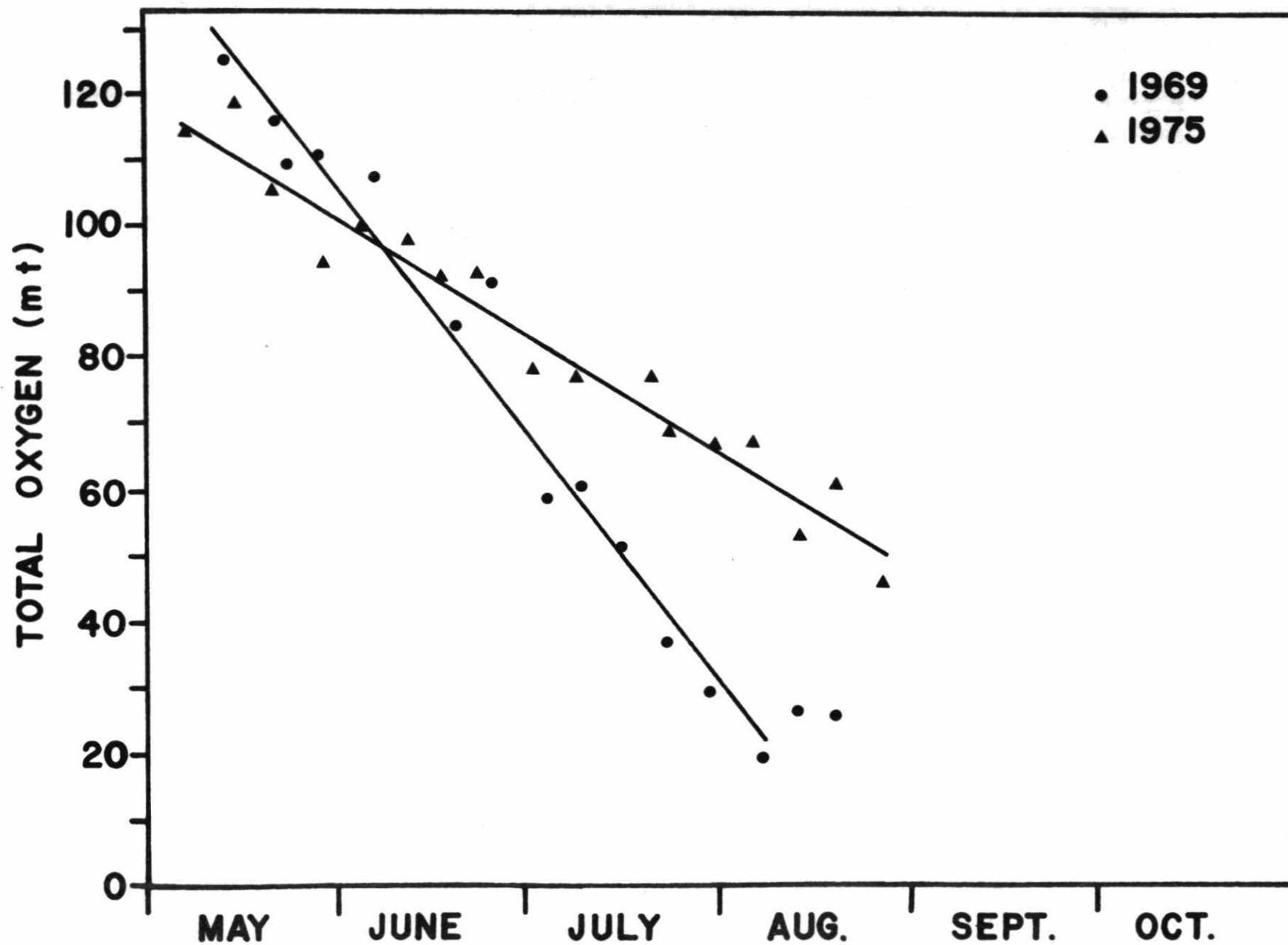


Fig. 3. Mass of oxygen in hypolimnion of Gravenhurst Bay as a function of time before (1969) and after (1975) P removal. Slope of the line is the areal hypolimnetic oxygen deficit rate.

Table 2: Comparison of trophic status parameters (defined in Table 1) measured in Gravenhurst Bay before P removal with those measured in 1975 and 1976 (in part, from Dillon et al. 1978).

Year	[P] (mg·m ⁻³)	(chl <u>a</u>) (mg·m ⁻³)	S.D. (m)	Phyto. (ASU·ml ⁻¹)	AHOD (mg·cm ⁻² ·d ⁻¹)
1969-71	44	9.8	2.5	2340	0.050
1975	20	5.0	3.9	1120	0.025
1976	27	10.6	3.7	4980	0.046

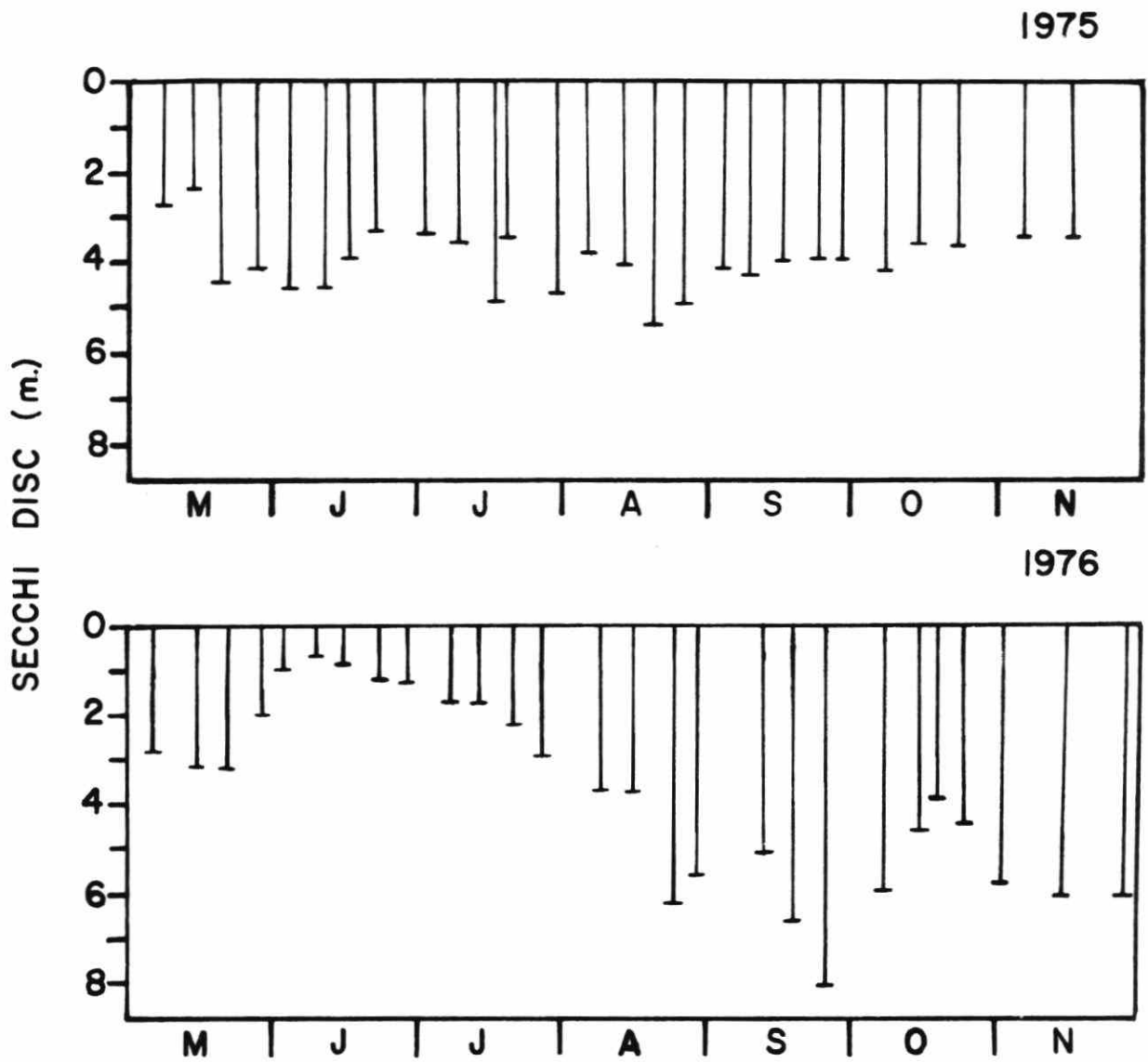


Fig. 4 Secchi disc depths in Gravenhurst Bay, 1975 and 1976.

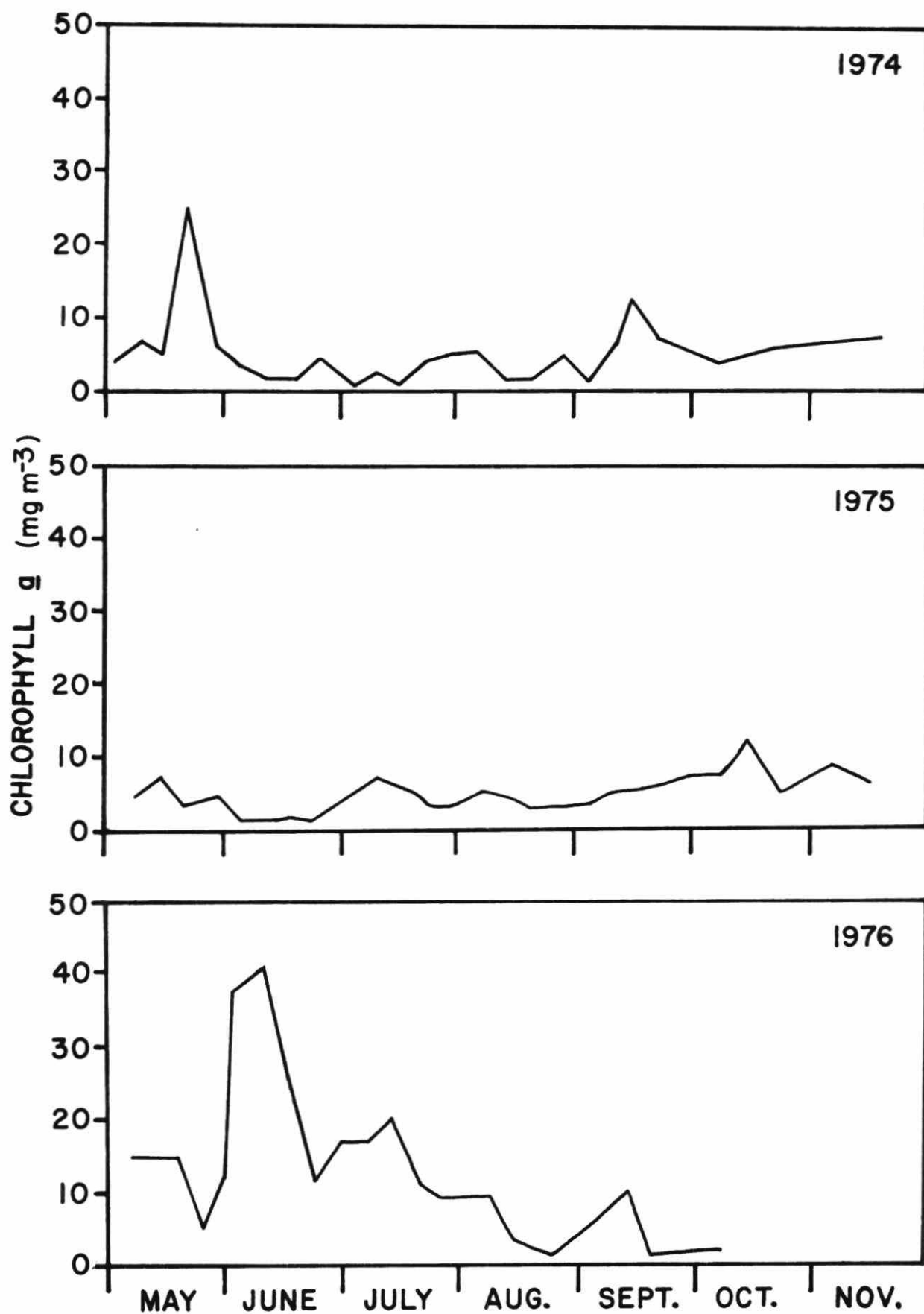


Fig. 5 Chlorophyll a concentration in the euphotic zone of Gravenhurst Bay (measured as twice the Secchi depth) from 1974-76.

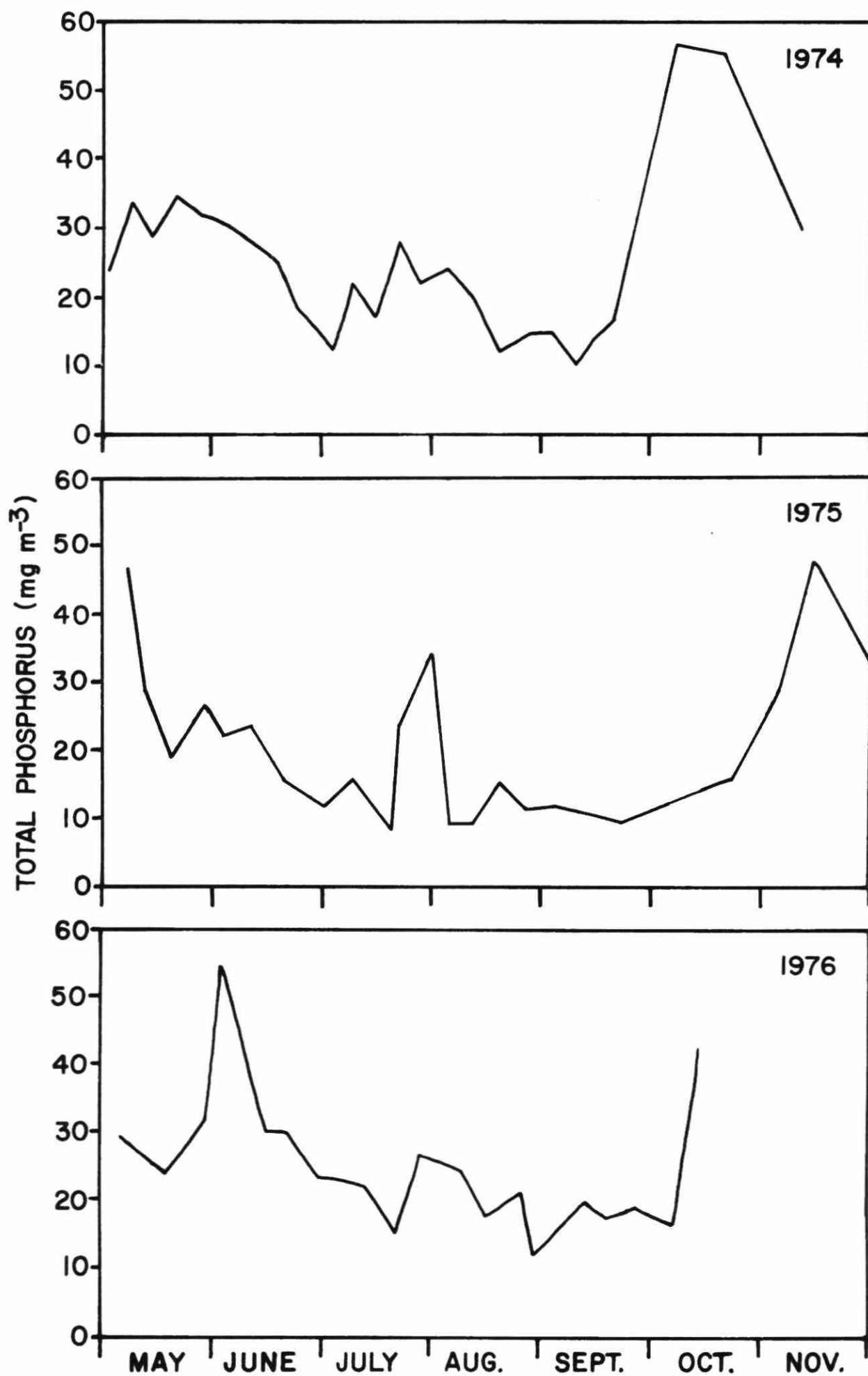


Fig. 6 Total phosphorus concentration in the euphotic zone of Gravenhurst Bay, 1974-76.

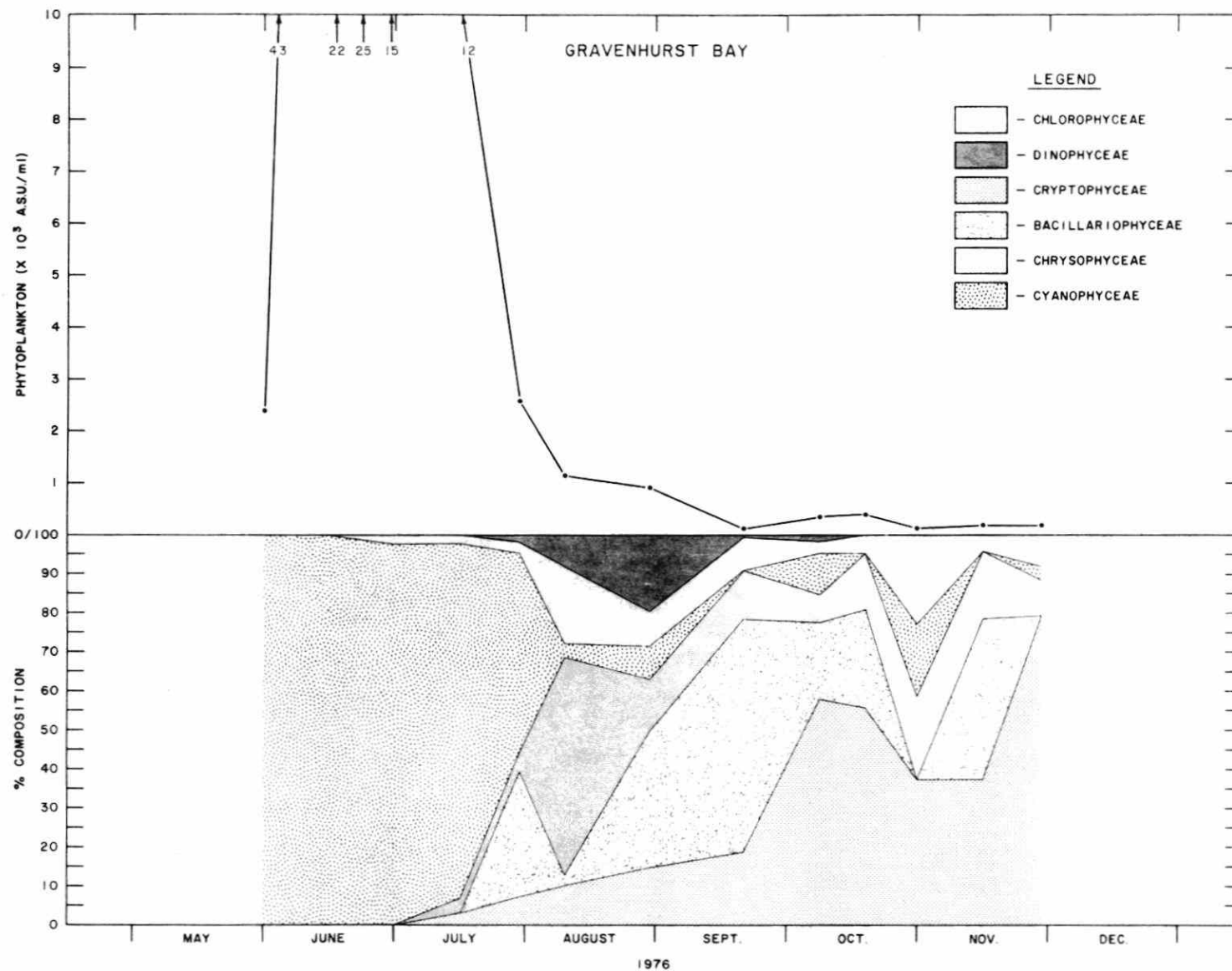
changed from one dominated by Aphanizomenon in June and July to one resembling oligotrophic Precambrian Shield lakes with low numbers of diatoms, motile Chrysophyceae and Cryptophyceae (Fig. 7).

Although changes since 1971 in phytoplankton biomass in Gravenhurst Bay have been dramatic and are relevant to many aspects of water quality (e.g. dissolved oxygen depletion - see Fig. 8), changes in phytoplankton composition are equally significant.

Declines in the Cryptophyceae and Chlorophyceae and increases in the Chrysophyceae have occurred independently of the large fluctuations of the Cyanophyceae or blue-green algae (Fig. 9). The blue-green algae, especially the bloom-forming genera, Aphanizomenon, Anabaena and Microcystis seem best able to respond quickly to changes in nutrient loading and were found in high densities during the summers of 1971 and 1976 as a consequence of high nutrient concentrations. This independence of the blue-green algae, especially the bloom-forming genera, from the rest of the algal flora is best illustrated by Fig. 9, which shows that the group is an important component of the plankton only when present at high density. During years when blue-green density is low, other algae are much more prominent in the plankton.

The blue-green algae have responded most to changes in total phosphorus loading and concentration; however, it would seem important to separate the bloom-formers (Aphanizomenon, Anabaena, Microcystis) from the "non-bloomers" (Aphanothece, Aphanocapsa, Merismopedia and Chroococcus) since the response of these two groups to the phosphorus control programme has been quite different. By 1975, the average blue-green "bloomer" biomass had declined approximately 78% to $170 \text{ A.S.U. ml}^{-1}$ from a pre-phosphorus control period average of $770 \text{ A.S.U. ml}^{-1}$. In contrast, the "non-bloomers" had increased from an average of $40 \text{ A.S.U. ml}^{-1}$ during the 1969-1971 period to $270 \text{ A.S.U. ml}^{-1}$ during 1975 (Fig. 10).

In other lakes in Ontario, certain Chrysophyceae (mainly species of Dinobryon, Uroglena, Mallomonas and Synura) are found in the metalimnion or upper hypolimnion of lakes along with several cyanophycean "non-bloomers" (e.g. Fig. 11 and unpub. data). In view of the similar niche occupied



Total density (Areal Standard Units/ml and composition of phytoplankton in Gravenhurst Bay during 1976. As a result of sewage treatment plant malfunction, the phytoplankton during the June-July period are indicative of extreme eutrophy. The rapid decline in algal density and alteration in composition following correction of sewage treatment plant malfunction, created typical oligotrophic characteristics in the plankton during the late summer and fall period.

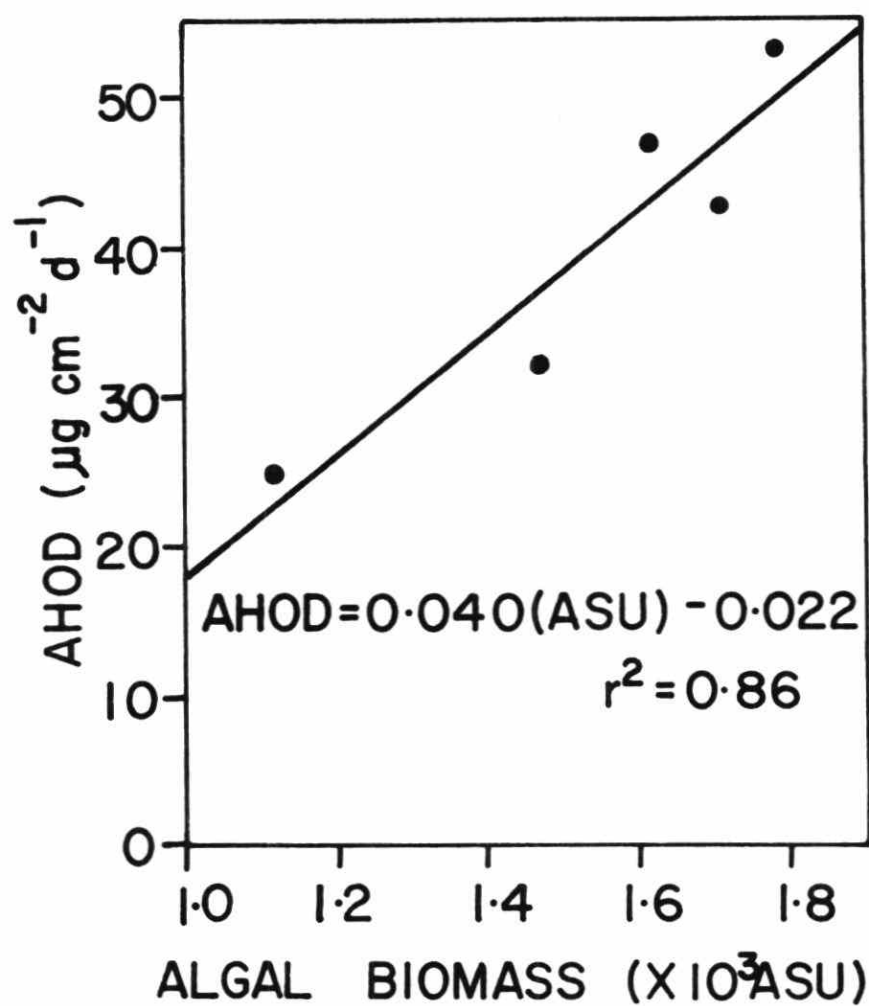


Fig. 8. Correlation between areal hypolimnetic oxygen deficit (AHOD) and average euphotic zone algal density (Areal Standard Units/ml) during a five year period in Gravenhurst Bay of Lake Muskoka.

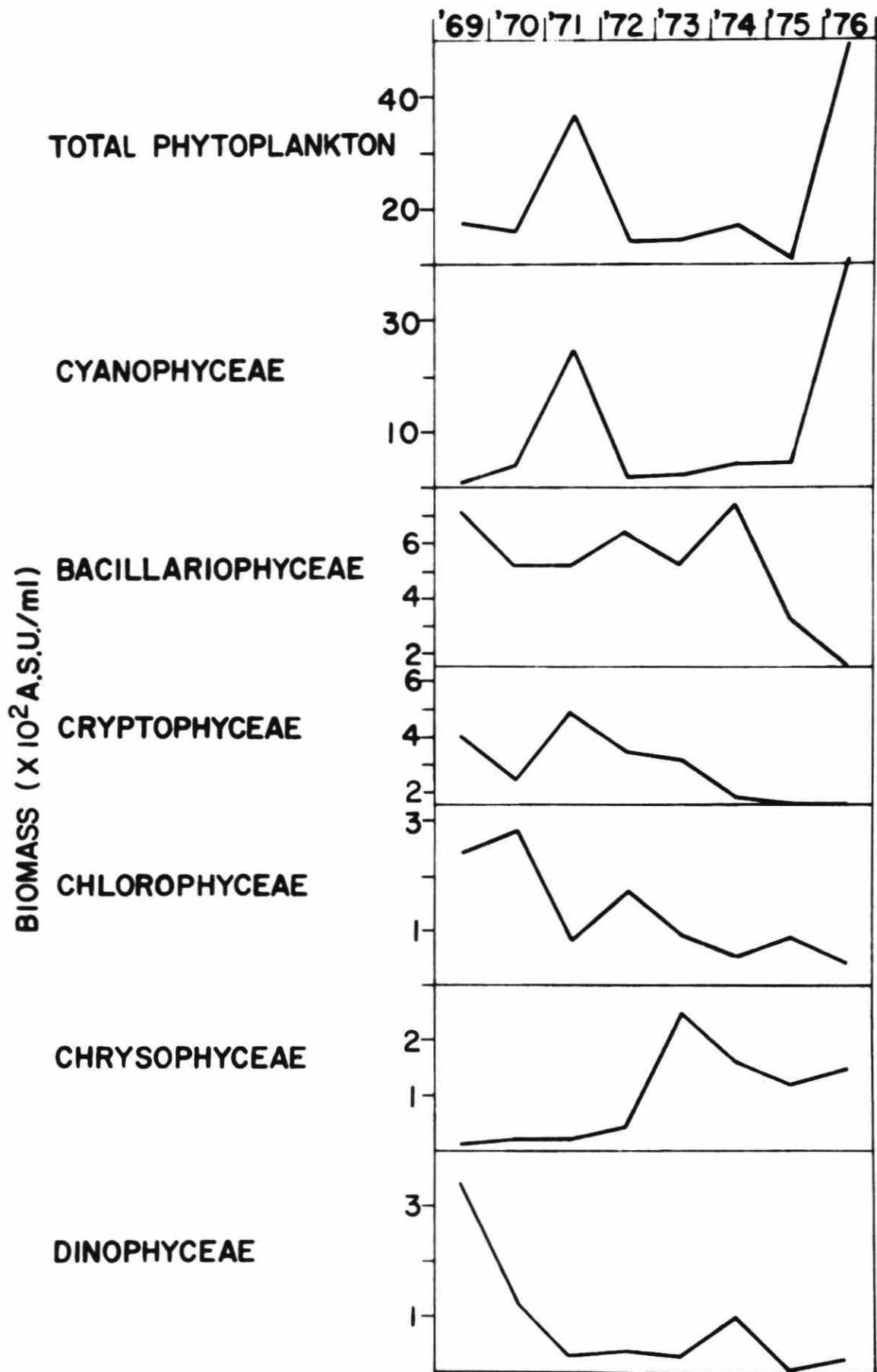


Fig. 9. Changes in average density of several important classes of algae in Gravenhurst Bay from 1969 through 1976.

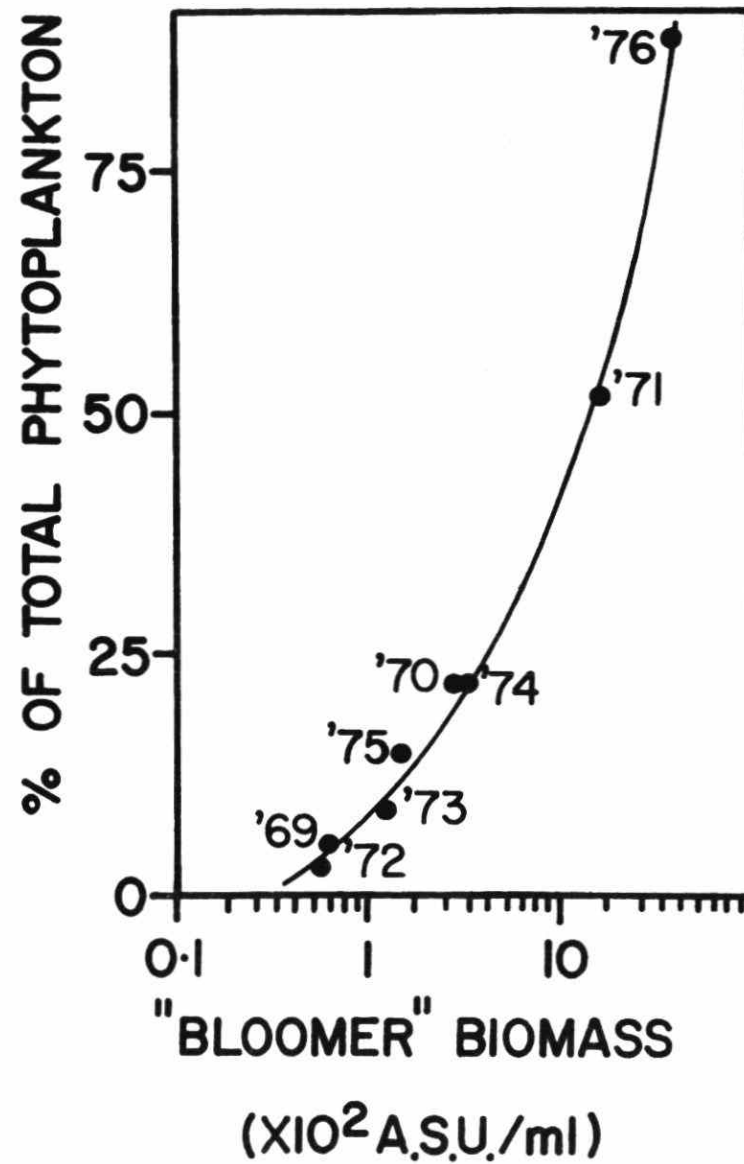
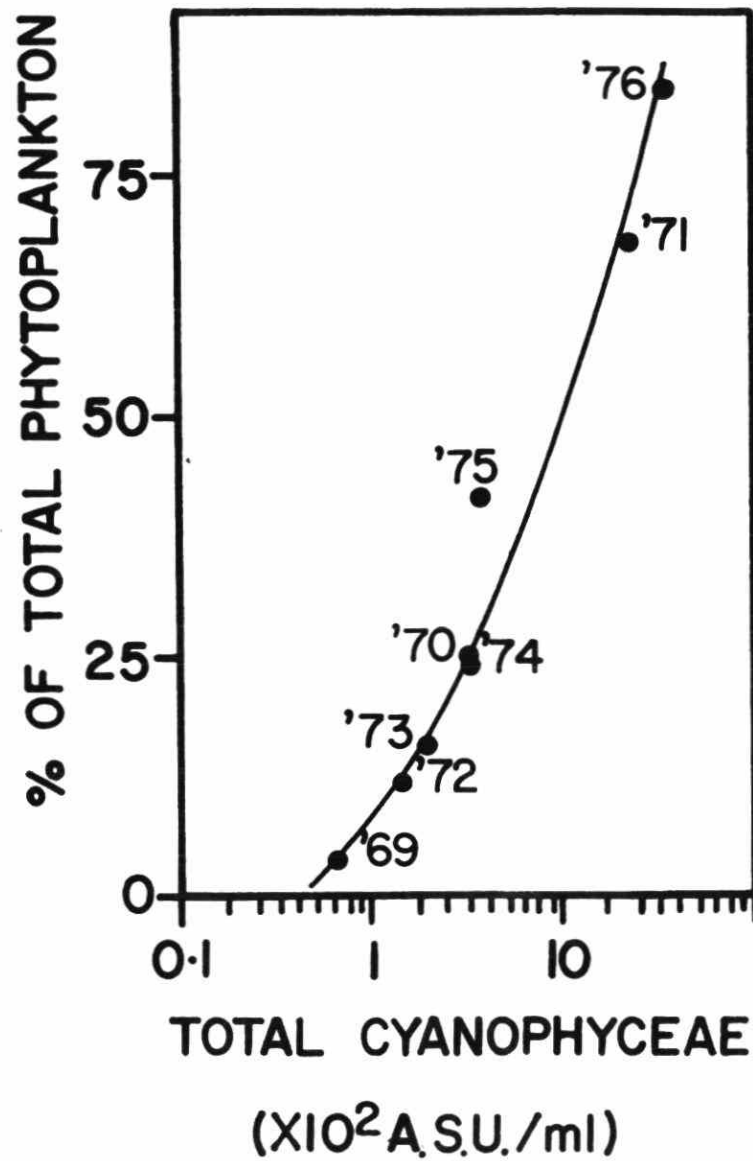


Fig. 10. Relationships between average total blue-green algal density (Cyanophyceae) and blue-green "bloomers" (i.e. mainly *Anabaena*, *Aphanizomenon* and *Microcystis* spp.) as a percentage of the total phytoplankton.

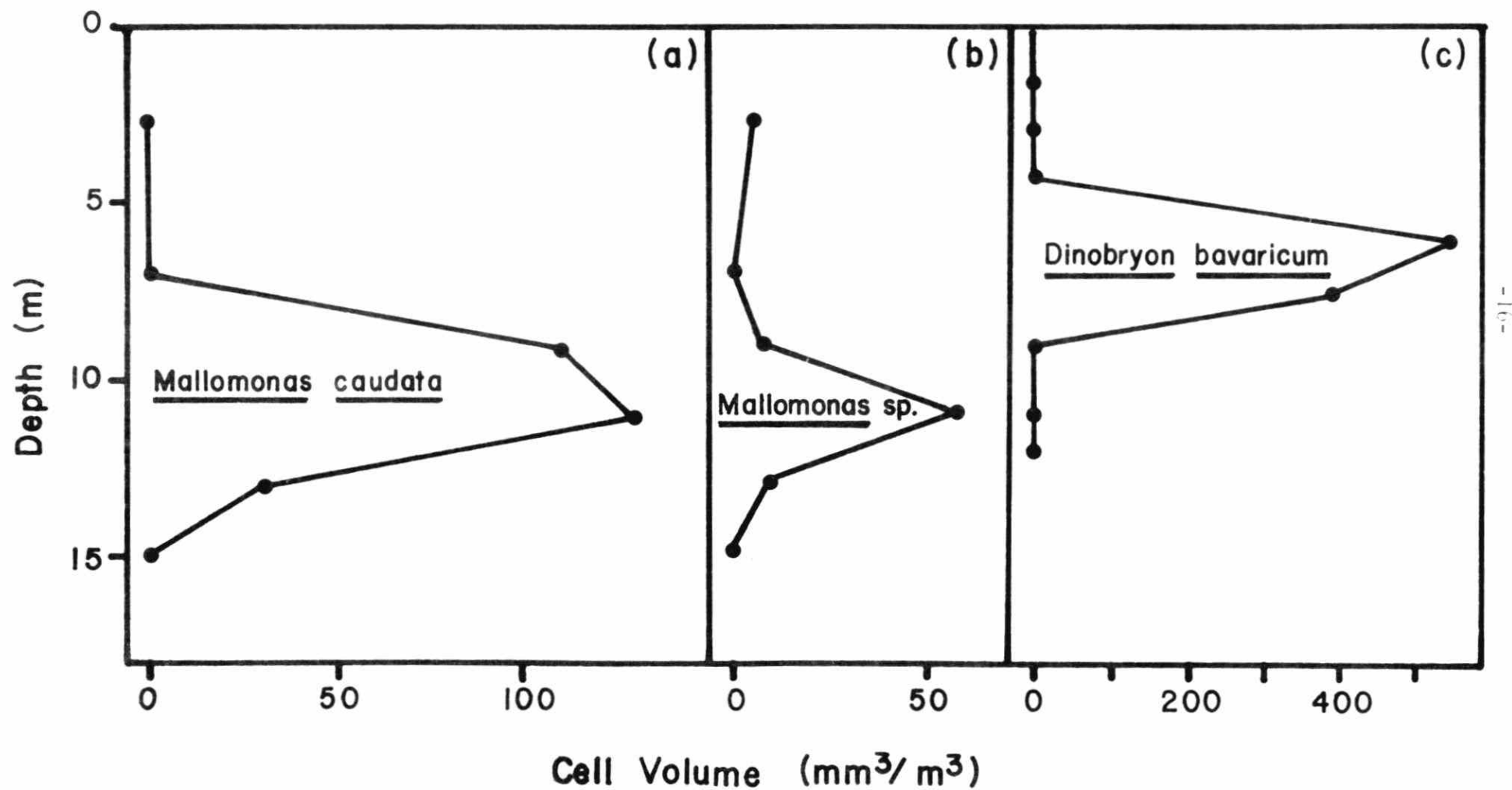


Fig. 11. Vertical distribution of some common members of the Chrysophyceae in (a) Buckskin Lake (78° 12' W, 45° 57' N), (b) Tadenac Lake (79° 57' W, 45° 03' N) and (c) Buchanan Lake (78° 50' W, 45° 17' N).

by both groups, it is not surprising that their responses to the change in nutrient input to Gravenhurst Bay would be similar. By 1975, the Chrysophyceae in the plankton of Gravenhurst Bay had increased to an ice-free period average of $120 \text{ A.S.U. ml}^{-1}$ from a pre-phosphorus control period average of only $16 \text{ A.S.U. ml}^{-1}$ (Fig. 12). Both the Chrysophyceae and the blue-green "non-bloomers" would appear to be particularly well adapted to low light and temperatures. Increases in biomass of these forms could only occur after significant declines in total phytoplankton (mainly diatoms and bloom-forming blue-green algae) in Gravenhurst Bay had allowed increased light penetration to the deeper waters, thereby allowing growth of forms more suited to these depths.

In years prior to 1972, there was little inorganic nitrogen in Gravenhurst Bay during the summer period (Fig. 13). Inorganic nitrogen built up as production declined while total nitrogen remained relatively constant over the entire season. The inorganic nitrogen disappeared in the following vernal period as it was incorporated into organic matter. In 1975, a year of P removal, the depletion of inorganic nitrogen did not occur over the summer period, indicating that it was not used by phytoplankton. This pattern was, however, reversed in 1976 after the sewage treatment plant failure and inorganic nitrogen distribution resembled pre-removal years (see also Table 3). Thus, response time to an increased input of P was very quick. Complete response to reduced input was also quick and appears to have taken about two months as can be seen from Fig. 14. The areal hypolimnetic oxygen demand plot can be broken down into two periods for 1976; up to the end of June ($0.088 \text{ mg O}_2 \text{ cm}^{-2} \text{ d}^{-1}$) and from July to August ($0.030 \text{ mg cm}^{-2} \text{ d}^{-1}$). This demonstrates the rapid response time generated by altering P inputs to lakes. Gilbertson, Dobson and Lee (1972) have shown a similar quick response time to nutrient loading and areal hypolimnetic oxygen demand in Lake Erie.

In summary, improvement in Gravenhurst Bay was proportional to the P reduction. Thus P removal can be critical in treatment of effluents to some waterways.

Middle Lake (Fig. 15) is a third case which has important qualitative as well as quantitative results. We have studied some of the acidic lakes

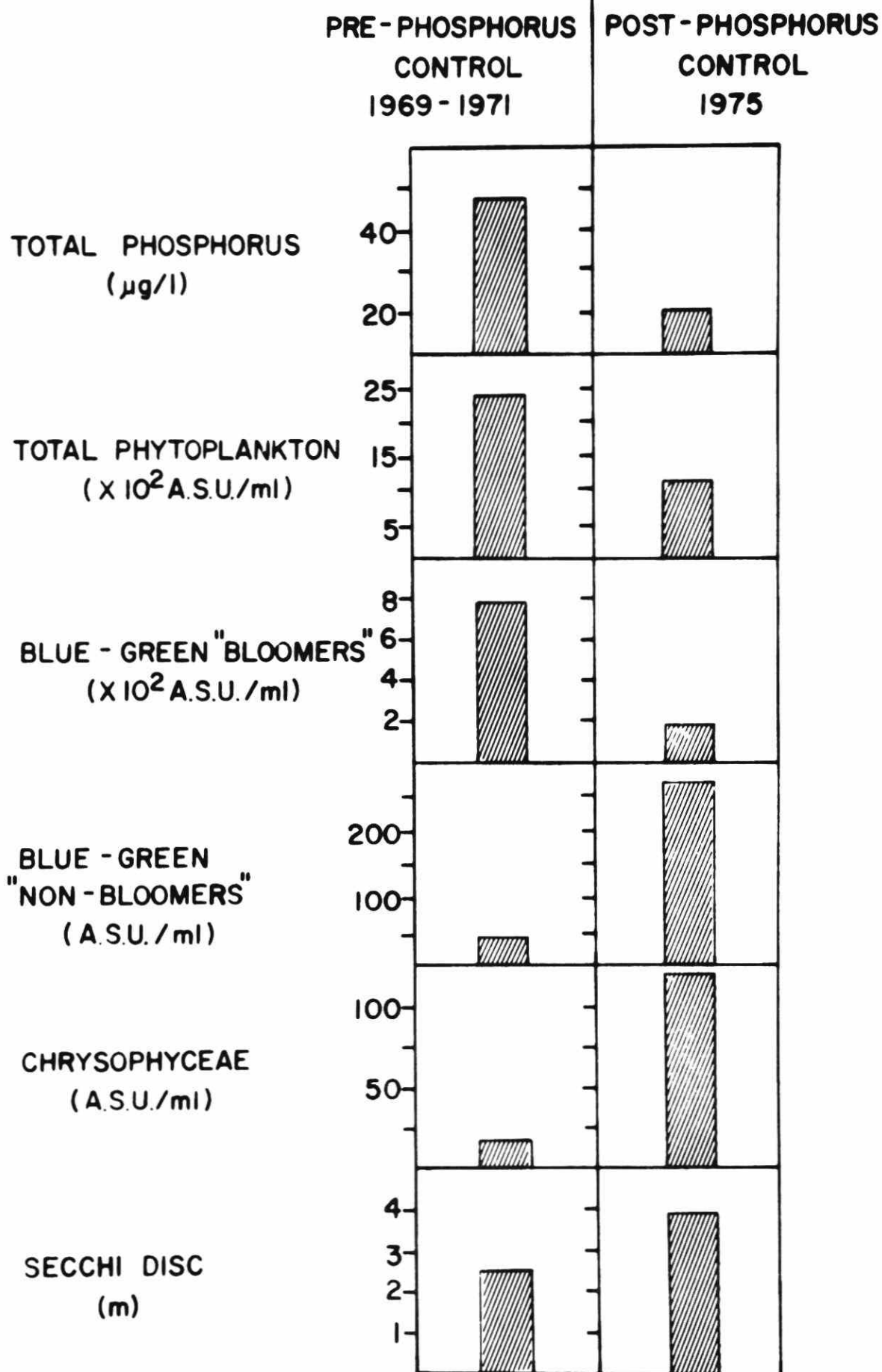


Fig. 12. Changes of some constituents of the Gravenhurst Bay phytoplankton following controls on phosphorus loading from the local sewage treatment plant.

Table 3: Changes in total phosphorus content and the partitioning of nitrogen between inorganic and organic forms in the euphotic zone of Gravenhurst Bay, 1969-76.

Year	[P] (mg.m ⁻³)	[TIN] (mg.m ⁻³)	[TON] (mg.m ⁻³)	TN/TP
1969	42	65	384	11.1
1970	39	77	427	13.9
1971	52	88	527	14.7
1972	35	144	383	17.8
1973	33	163	399	18.3
1974	25	188	356	24.8
1975	20	257	320	35.2
1976				
April-Aug.	29	70	580	23.6
Aug.-Nov.	27	180	270	19.3

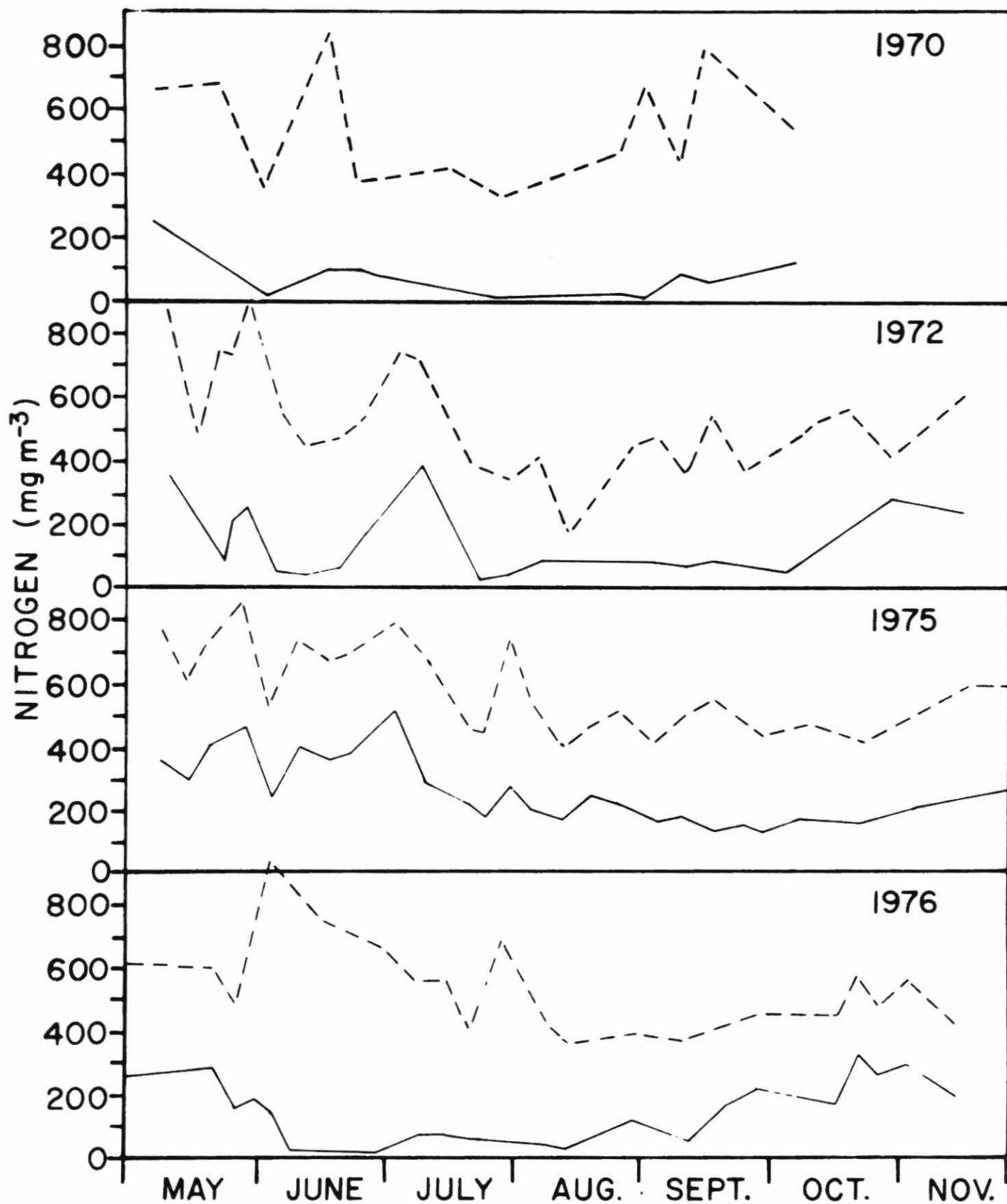


Fig. 13 Total inorganic nitrogen (TIN = $\text{NH}_3 + \text{NO}_2 + \text{NO}_3$; solid line) and Total nitrogen (TN = TIN + organic nitrogen; dotted line) in Gravenhurst Bay prior to phosphorus removal (1970), 1 and 4 years after removal was initiated (1972 and 1975 respectively), and following a short-term increase in phosphorus load (1976).

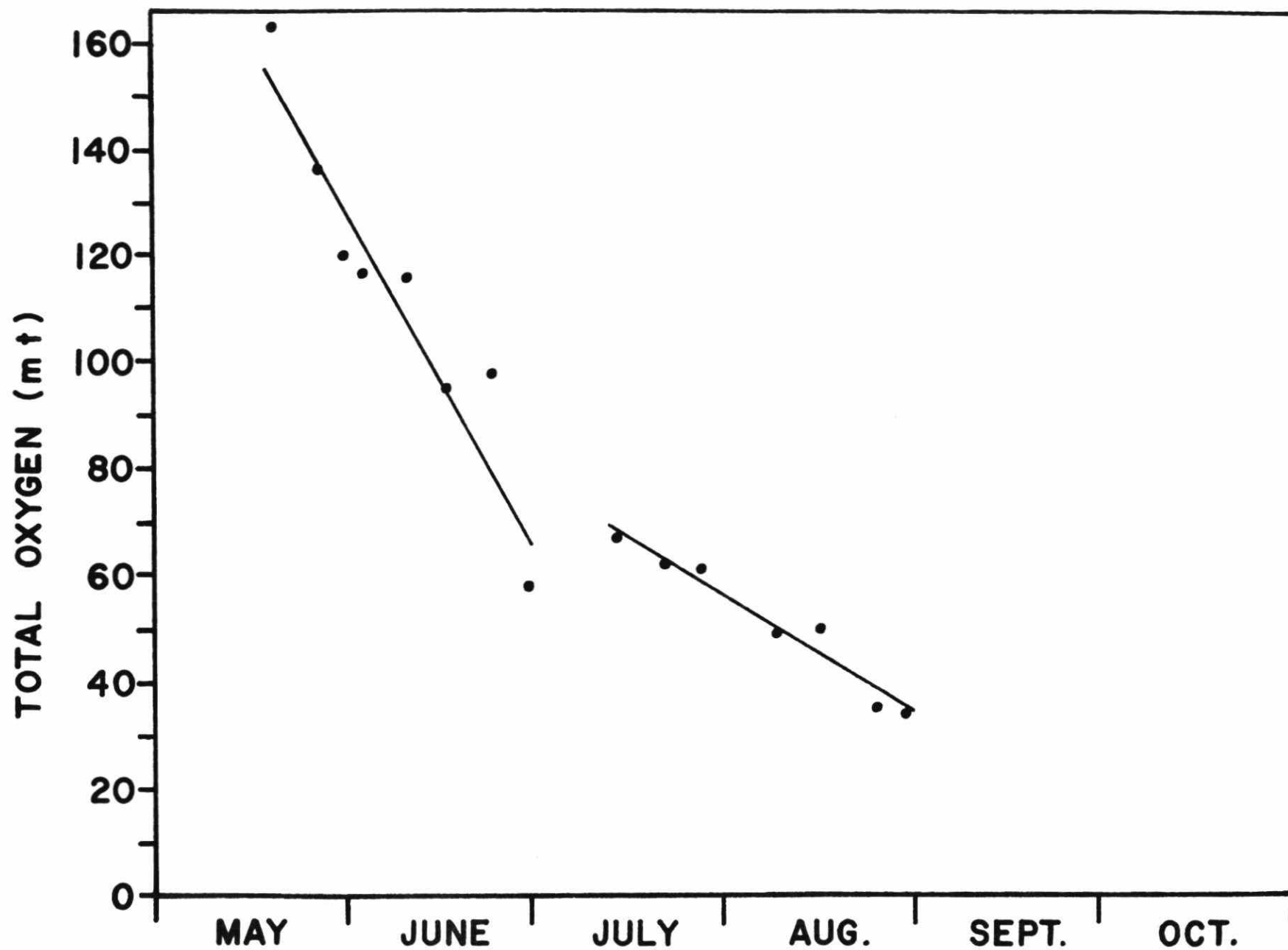


Fig. 14. Mass of oxygen in hypolimnion of Gravenhurst Bay in 1976 as a function of time. Rate of decrease was high in May and June, much slower in July and August.

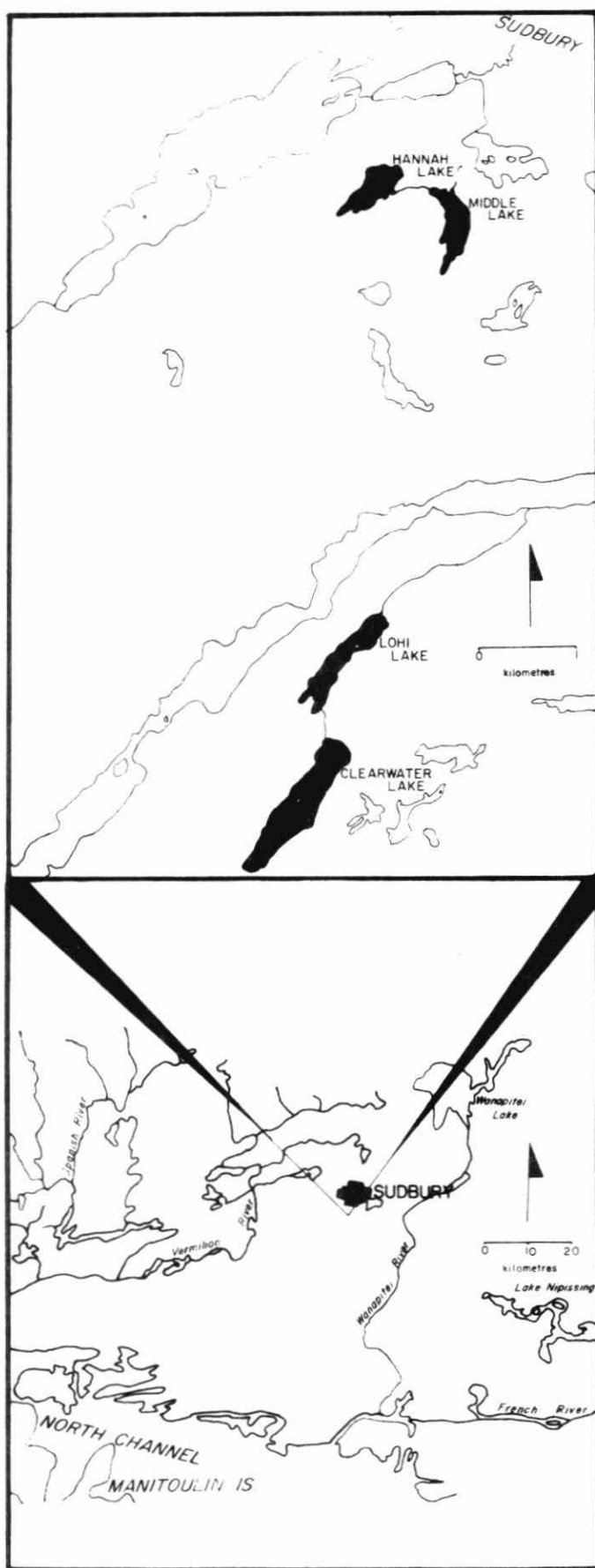


Fig. 15 Location of Middle Lake in the Sudbury region.

near Sudbury now for the last four years with the intent of understanding the biological and chemical responses of these lakes to various chemical treatments, including decreased and increased pH as well as nutrient additions. In 1973 Middle Lake had a pH of 4.2 which was raised to approximately 7 by addition of $\text{Ca}(\text{OH})_2$ and CaCO_3 (Fig. 16). Middle Lake was fertilized in 1975 with P alone and this raised the P concentration from something near the detection limit of $2 \mu\text{g l}^{-1}$ to $9 \mu\text{g l}^{-1}$. This new 'high' post-fertilized level is still very low when compared to P concentrations in oligotrophic Precambrian lakes.

Although there was a quantitative biological response due to the fertilization of Middle Lake (e.g. chlorophyll a and phytoplankton biomass increased) (Table 4), the significant change was qualitative. The phytoplankton population in 1973 at pH 4.2 was dominated by cryptophytes, dinoflagellates, chrysophytes and chlorophytes (Fig. 17) which is typical of acid lakes in Sweden (Hörnström et al. 1973) and Ontario (Yan et al. 1977). In 1974, at a pH of 6.5 to 7.0, the algal composition was altered drastically so that chrysophytes now dominated the phytoplankton community with diatoms appearing at different time intervals (Fig. 17), a seasonal pattern typical of circumneutral oligotrophic Precambrian lakes. Following P additions in late June of 1975, the community again drastically changed with blue-green algae completely dominating. Also during this time pH never exceeded 7, which is contrary to the theory of Shapiro 1973 that low pH excludes blue-green algae in lakes. Thus, this experiment clearly indicated the importance of P, even at what are normally considered to be very low concentrations.

Preliminary data from another acid lake, Mountaintop Lake, also in the Sudbury area (pH = 4.2), suggest that unless reserves of inorganic carbon are increased artificially as they were in Middle Lake by neutralization with $\text{Ca}(\text{OH})_2$ and CaCO_3 , the expected yield of phytoplankton from additions of phosphorus are not achieved as quickly (nitrogen as well as phosphorus has been added to Mountaintop Lake). An average concentration of phosphorus (supplied as H_3PO_4) of about $60 \mu\text{g P l}^{-1}$ was achieved by fertilization. However, after the first year, only about 15% of the expected yield of $9\text{-}10 \text{ mm}^3 \text{ l}^{-1}$ phytoplankton was produced. We suspect that this lake is carbon limited under a regime of such high rates of nitrogen and phosphorus additions.

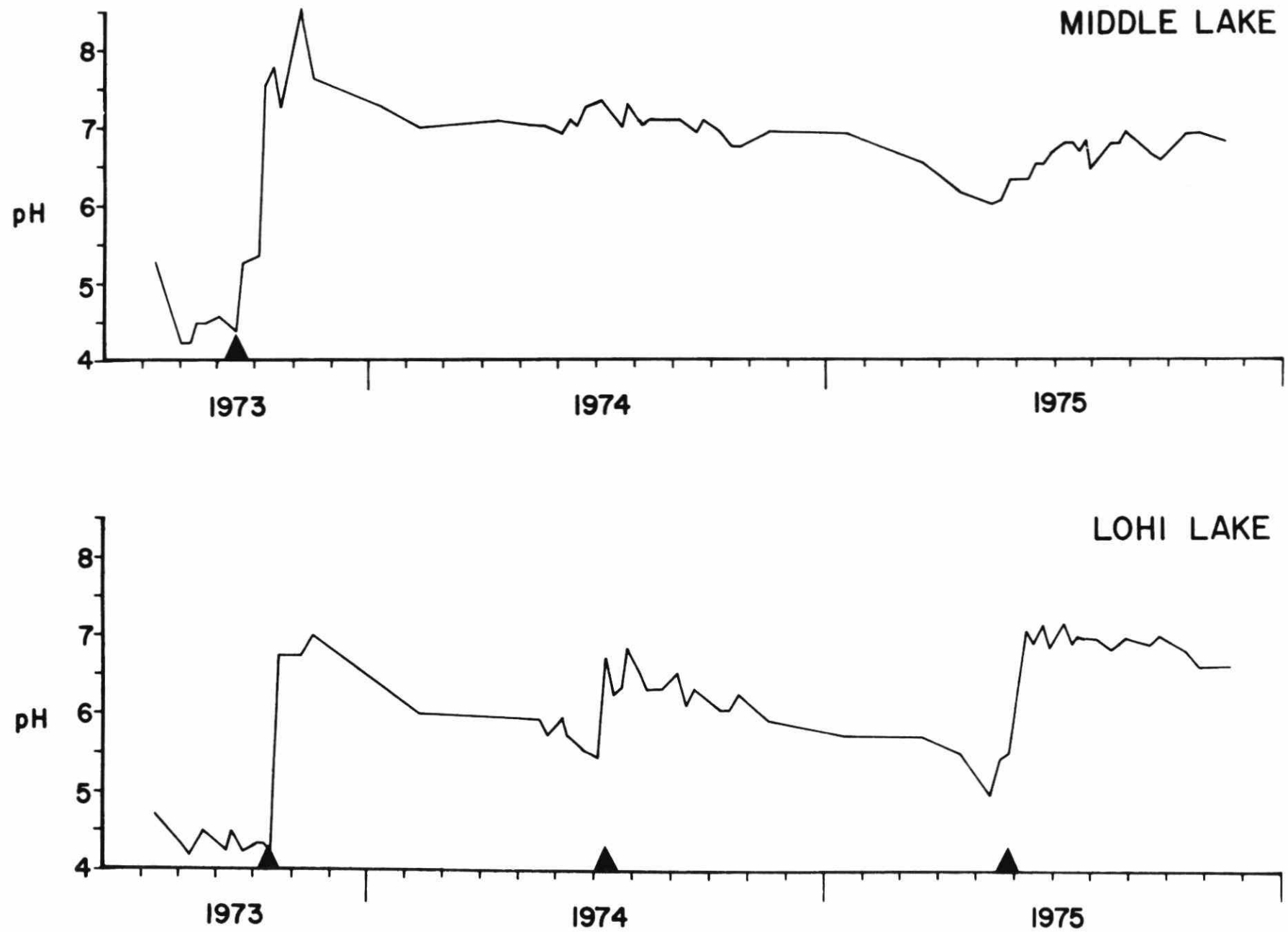


Fig. 16. pH of Middle and Hannah Lakes, 1973-75. Triangles denote application of $\text{Ca}(\text{OH})_2$ and CaCO_3 .

Table 4: Demonstration of the effects of fertilization on Middle Lake
(from Scheider and Dillon 1976).

	Pre- fertilization	Pre- fertilization
Total phosphorus ($\text{mg} \cdot \text{m}^{-3}$)	2.5	9.0
chlorophyll <u>a</u> ($\text{mg} \cdot \text{m}^{-3}$)	1.1	2.4
Phytoplankton (A.S.U. $\cdot \text{ml}^{-1}$)	143	703
Zooplankton (numbers $\cdot \text{l}^{-1}$)	0.64	2.32
Benthos (numbers $\cdot \text{m}^{-2}$)	96	138

MIDDLE LAKE

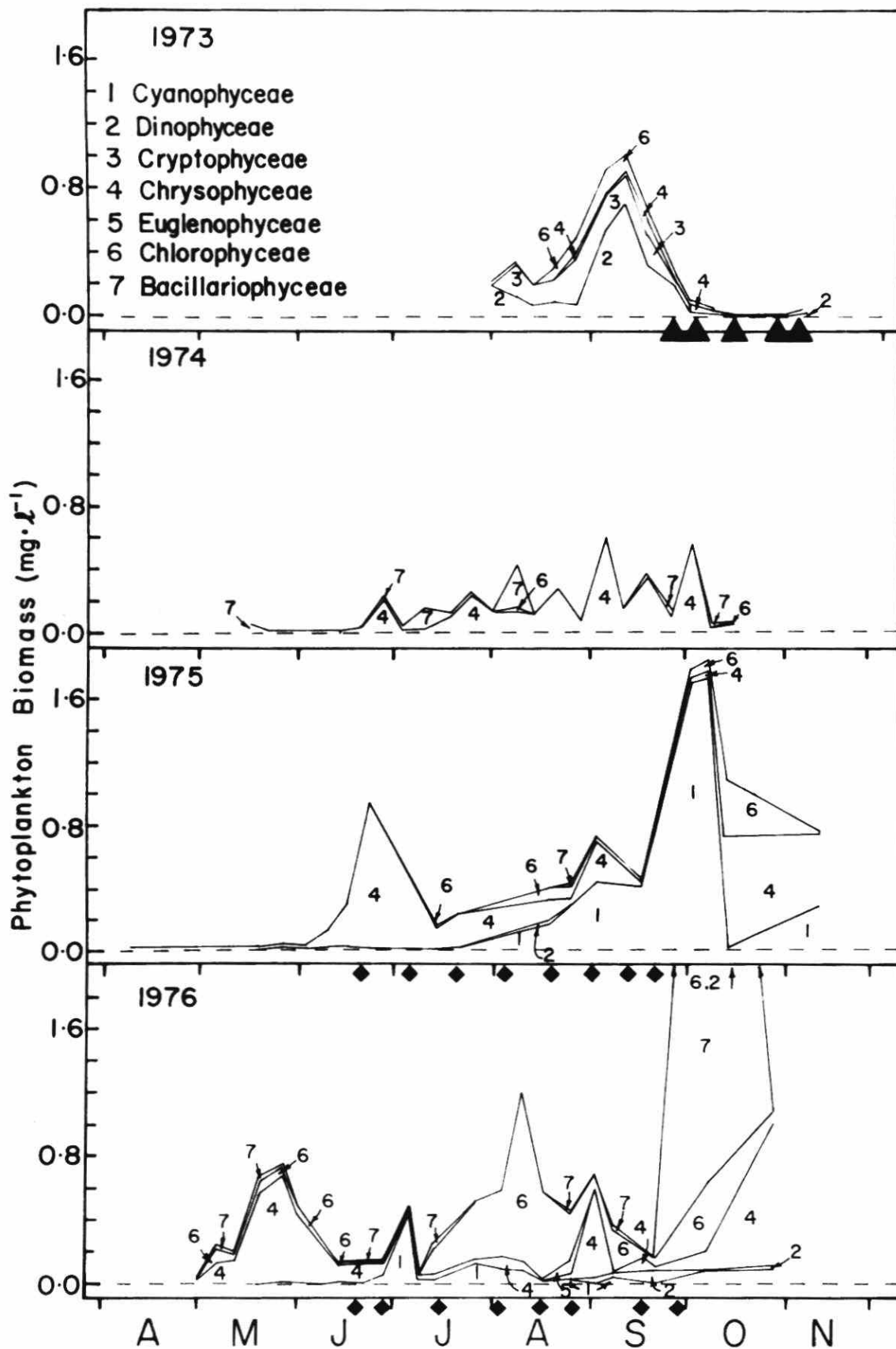


Fig. 17. Phytoplankton biomass ($\text{mg} \cdot \text{l}^{-1}$) and composition by major classes in Middle Lake, 1973-76.

It is worth emphasizing that Ontario Lakes show a well defined relationship between the average biomass of phytoplankton during the ice-free period and the average total phosphorus concentration (e.g. see Fig. 45). Given little year-to-year change in nutrient concentration, seasonal distribution of phytoplankton would also seem to be remarkably constant from year-to-year (Fig. 18). For many lake locations where we have several years of data, it is apparent that little year-to-year change in phosphorus concentration results in little change in average phytoplankton biomass (Fig. 19). In contrast, in those lakes which do experience a change in phosphorus concentration resulting either from lake fertilization or from improved waste treatment and decreased phosphorus loading, the resultant change in phytoplankton yield can be predicted from the empirical data (Fig. 19).

The next point to be made is that the parameter of importance is the P concentration in the lake, not the P loading. Vollenweider's original plot (1969) and its subsequent alteration (Vollenweider 1975, 1976) to L versus \bar{Z}/τ_w (Fig. 20) both have lines separating lakes into oligotrophic, mesotrophic and eutrophic categories based on a subjective assessment of lake conditions. The second plot, or one of its later variations, has definitely yielded a more realistic representation by including water replenishment time (τ_w) or its inverse, flushing rate (ρ) as shown in a study of Cameron Lake and Four Mile Lake in southern Ontario (Table 5) (Dillon 1975). Both lakes were of similar morphometry but Cameron Lake had an areal P loading (L) twenty times that of Four Mile Lake. The P and chlorophyll *a* concentrations and the Secchi disc depths, however, were virtually identical. The reason for the difference was that a high flushing rate in Cameron Lake offset the high loading rate. Thus, not only does the L versus \bar{Z}/τ_w plot rationally give a better fit to empirical data, as illustrated by the above case, but the lines separating the lake classes have slopes with appropriate units of $\text{mg}\cdot\text{m}^{-2}\cdot\text{yr}^{-1}/\text{m}\cdot\text{yr}^{-1}$ or $\text{mg}\cdot\text{m}^{-3}$. Therefore, both the rate and areal concepts are removed from consideration and lakes are separated according to some simple measure of their P concentration.

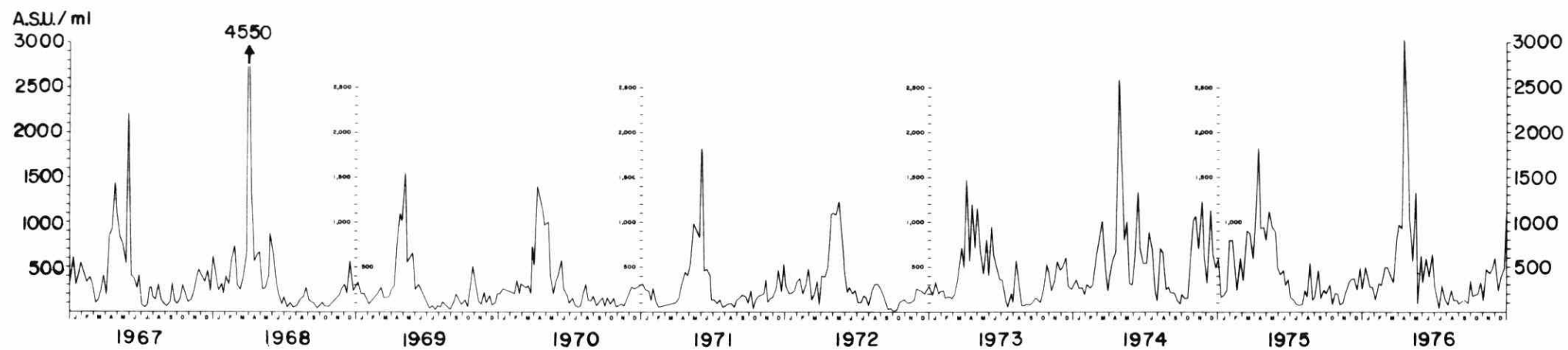


Fig. 18. Seasonal changes in total phytoplankton density at the outflow of Lake Ontario (Brockville) over a 10 year period.

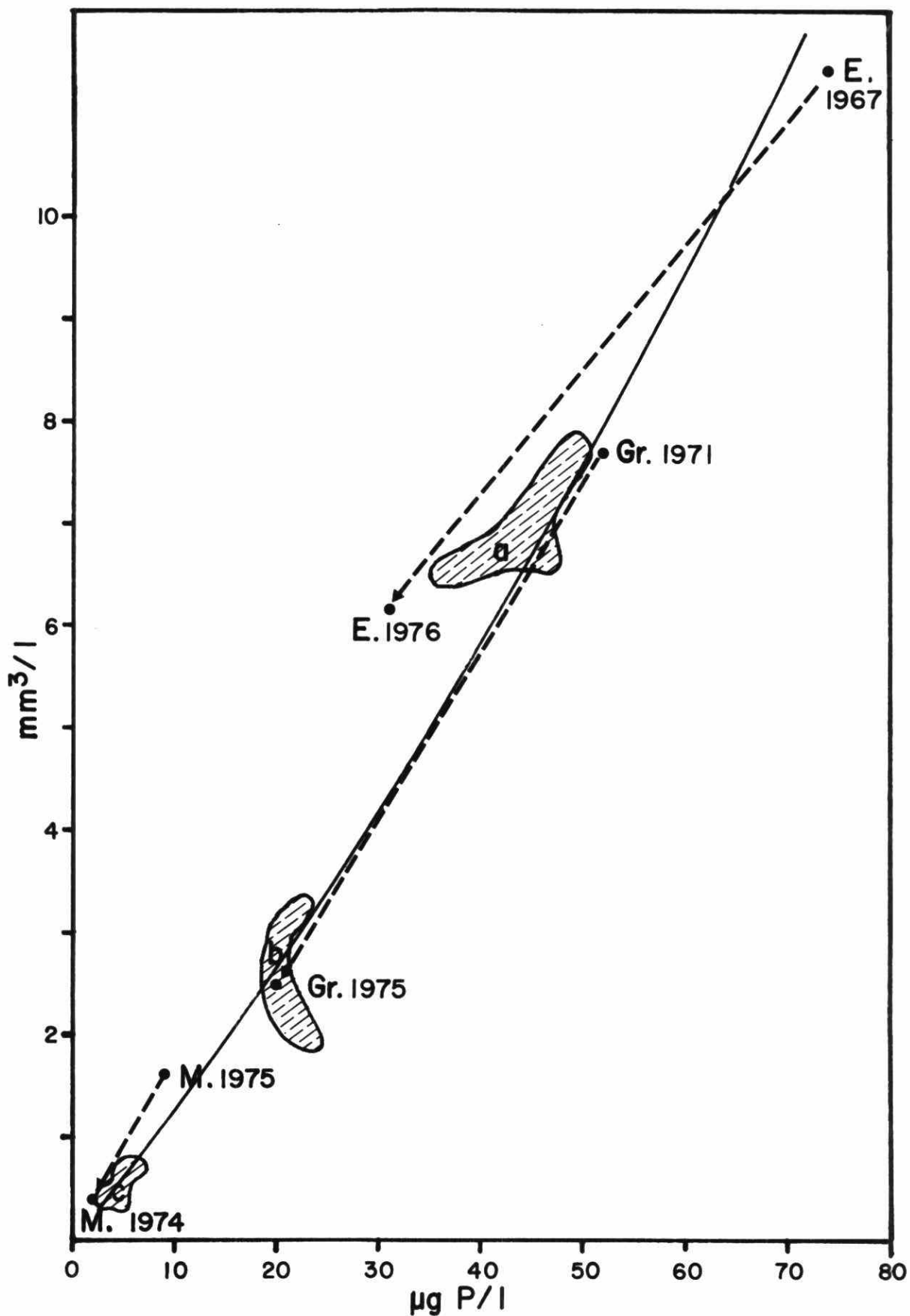


Fig. 19. Relationship between average total phosphorus concentration and average phytoplankton biomass (mm^3/l) at several lake locations in Ontario. Keyed as follows: E, western Lake Erie; Gr, Gravenhurst Bay of Lake Muskoka; M, Middle Lake near Sudbury (fertilized with H_3PO_4); a, 5 years of data from the Bay of Quinte off Picton; b, 5 years of data from the Bay of Quinte off Conway; c, 5 years of data from Clearwater Lake near Sudbury.

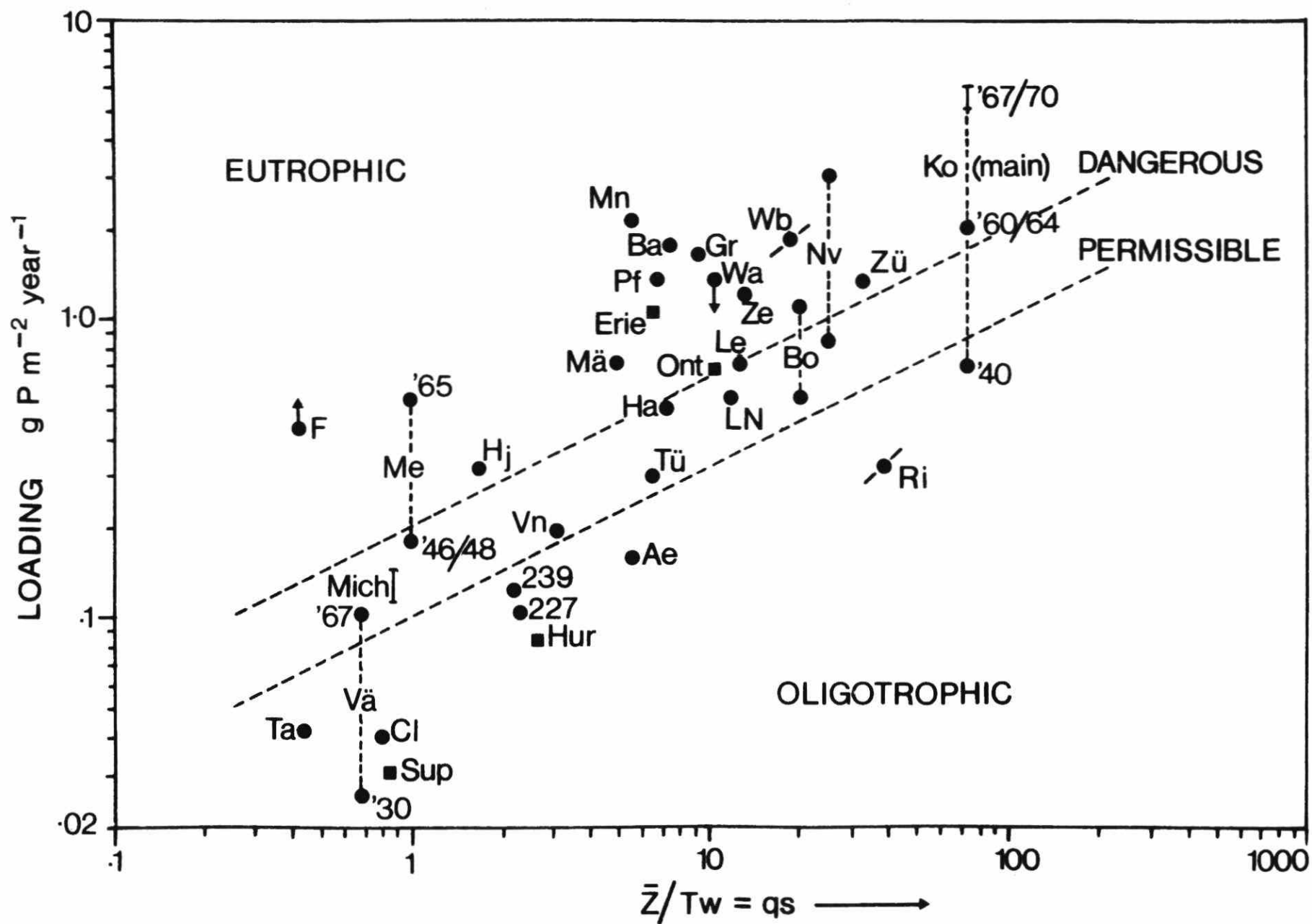


Fig. 20. Phosphorus load vs hydraulic load (q_s). Lines designating "permissible" and "dangerous" loading levels as a function of q_s are denoted. For further discussions, see Vollenweider (1975, 1976).

Table 5: Demonstration of the modifying effect of flushing rate on phosphorus loading. Two lakes of similar morphometry, one with a phosphorus load twenty times greater than the other, have similar lake phosphorus concentrations because of differences in flushing rates.

Parameter		Cameron	Four Mile
A_0	(km^2)	12.7	7.7
\bar{z}	(m)	7.1	9.3
L_p	($\text{g} \cdot \text{m}^{-2} \cdot \text{yr}^{-1}$)	2.21	0.11
[P]	($\text{mg} \cdot \text{m}^{-3}$)	10	9
[chl <u>a</u>]	($\text{mg} \cdot \text{m}^{-3}$)	1.6	1.4
S.D.	(m)	4.3	5.5
ρ	(yr^{-1})	16	0.24

It is not surprising that P concentration should correlate so well with the subjective assessment of a lake's trophic status since this assessment is normally based on parameters that reflect standing crop (e.g. chlorophyll a or algal biomass) which are also in units of concentration. If they had been based on rate processes such as primary production then perhaps a P input rate would have been a more appropriate parameter.

The linkages or sequence of factors affecting P concentration in a lake now appear as in Fig. 21 with load being just one of the factors controlling P concentrations. This can be expanded into a more detailed set of interrelationships (Fig. 22) which is still analagous to Fig. 17 with its three major groupings. Trophic status, under this new procedure, is defined by measuring the parameters of chlorophyll a, algal biomass, Secchi disc and areal hypolimnetic oxygen demand. P concentration is represented by the concentration at spring turnover which, in turn, is derived from the P loading from a variety of sources and is modified by morphometry and hydrology.

This sequence is the strategy being currently used by the Lakeshore Capacity Study within the Ministry of the Environment for arriving at answers to the question presented in the introduction. The sequence is outlined in the following pages.

From studies of watersheds in southern Ontario, combined with a literature survey, it became apparent that bedrock geology and land use were the two major factors controlling P yield or export from watersheds. From Dillon's and Kirchner's (1975) results for the two predominant bedrock types in Ontario combined with two types of land use (Table 6), the disconcerting feature is the broadness of the ranges determined for P export. Kirchner (1975) tried to explain some of this variation for forested igneous watersheds and found that drainage density (D_d = stream length/drainage area) explained most of the variation (Fig. 23). From this it is possible to make reasonably accurate predictions of the P export for a large number of lakes.

The second source of P is dry or wet precipitation. From Lakeshore Capacity Study data and the literature, a range of values of $30 - 70 \text{ mg} \cdot \text{m}^{-2} \cdot \text{yr}^{-1}$

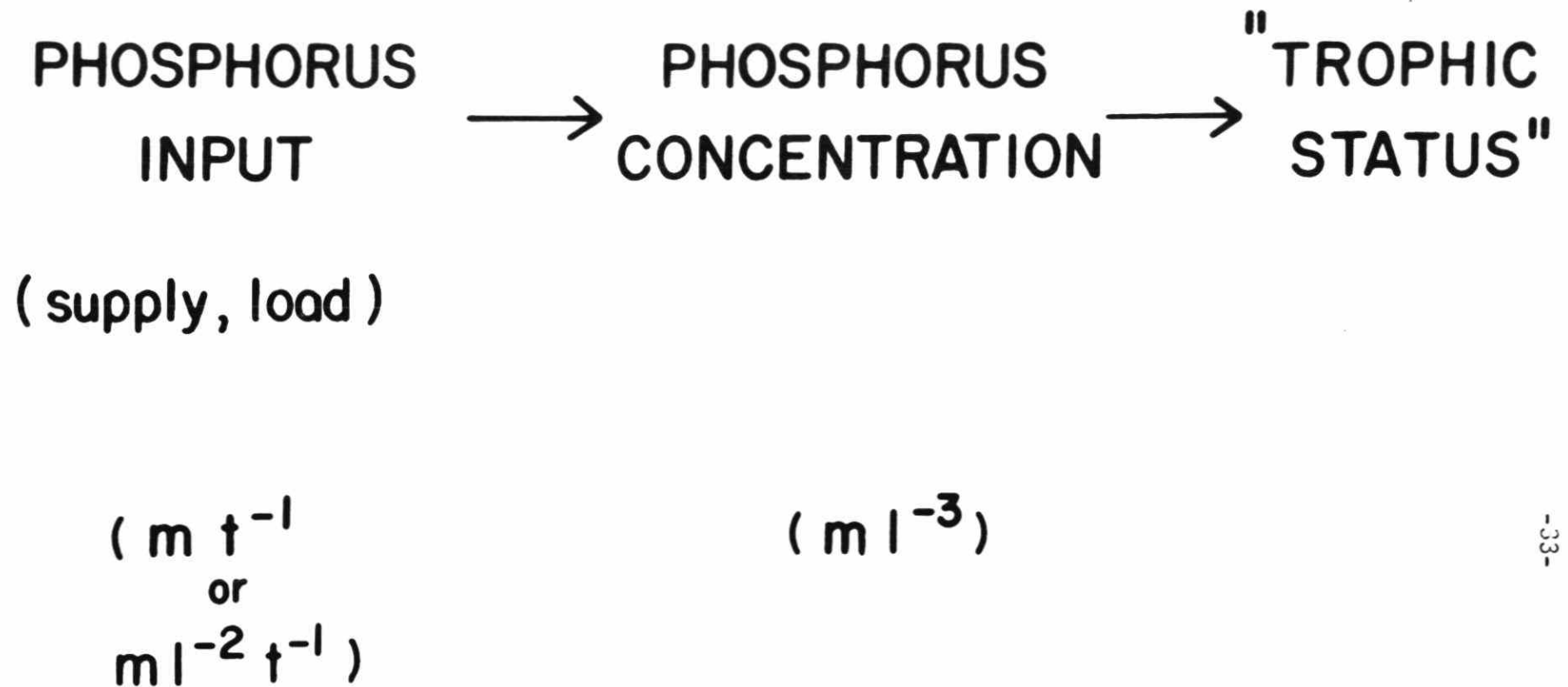


Fig. 21. Linkages affecting phosphorus concentration
(m = mass, l = length, t = time).

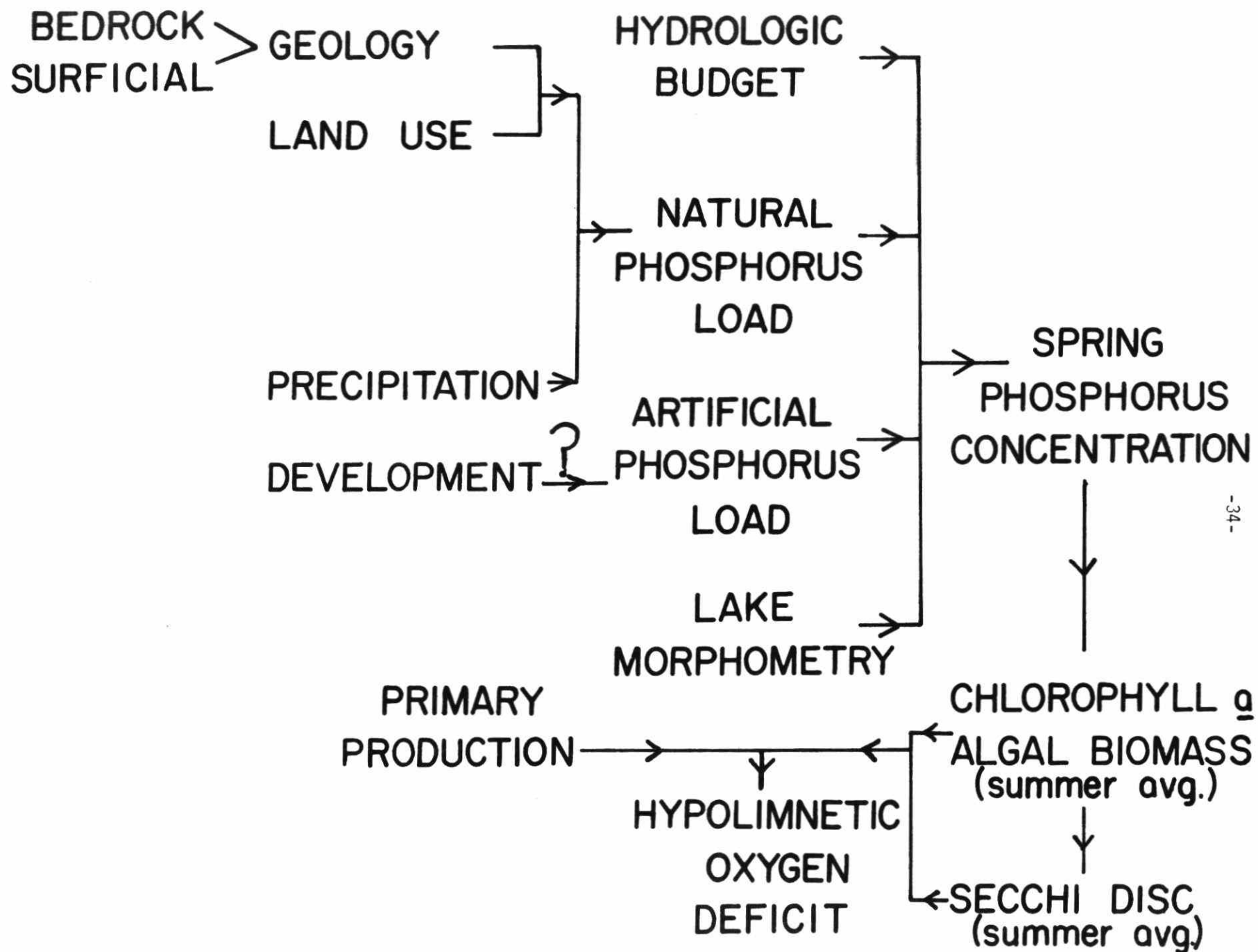


Fig. 22. Detailed outline of linkages between watershed and water quality of a lake (from Dillon and Rigler 1975).

Table 6: Ranges and mean values for export of total phosphorus from 43 watersheds. Values include results from present study. Results in $\text{mg}\cdot\text{m}^{-2}\cdot\text{yr}^{-1}$ (from Dillon and Kirchner 1975).

Land Use	Geological Classification	
	Igneous	Sedimentary
Forest		
Range	0.7 - 8.8	6.7 - 18.3
Mean	4.7	11.7
Forest + Pasture		
Range	5.9 - 16.0	11.1 - 37.0
Mean	10.2	23.3

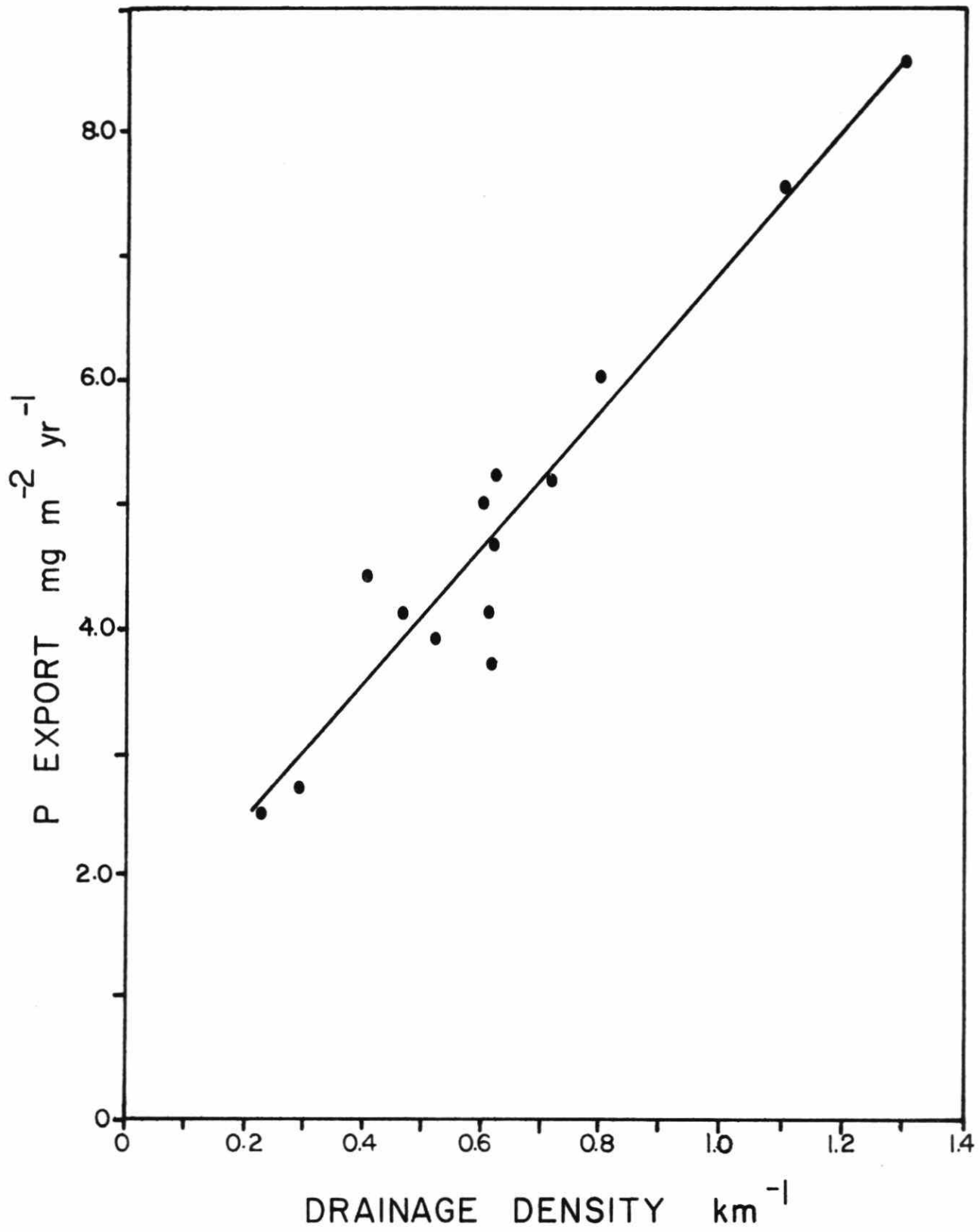


Fig. 23 Phosphorus export as a function of drainage density (from Kirchner 1976).

has been established. An average of $50 \text{ mg P m}^{-2}\text{yr}^{-1}$ is a fairly reliable estimate to use for aeolian loading onto a lake's surface.

In the past, Dillon and Rigler (1975) estimated the P input from human sources by applying $0.8 \text{ kg capita}^{-1} \text{ yr}^{-1}$ to give a possible annual P contribution. This can be qualified, depending on the kind of effluent treatment practiced, or in the case of septic tank, tile field systems, modified by a soil retention factor (Table 7) as measured by Brandes et al. (1974). The problem here is that retention factors for very few soil types have been measured. Most apply to artificial mixtures and represent only the tile field and filter bed. In fact, determining the P input from development in situations where septic tanks and not sewage treatment plants exist is one of the purposes of the Lakeshore Capacity Study.

That this problem is complex was demonstrated by Nicholls (1976) in a study of Harp and Jerry Lakes on the Precambrian Shield. Harp Lake has 73 shoreline cottages while Jerry Lake has none. Both lakes are not only morphometrically alike, but also similar in their measurements of parameters defining trophic status (chlorophyll a, Secchi disc, total organic nitrogen, areal hypolimnetic oxygen demand, winter oxygen demand) (Table 8). The data seem to implicate that the 73 cottages on Harp Lake have had little impact on the lake in comparison to Jerry Lake but there are qualitative differences that suggest otherwise. Although algal biomasses were similar in the two lakes, the blue-green algae were more important than diatoms in Harp Lake, while the reverse is true in Jerry Lake. Furthermore, the most important diatom in Harp Lake was Tabellaria fenestrata, a species that has preceded or accompanied nutrient enrichment problems in a number of cases. The same species was entirely absent in Jerry Lake, where Rhizoselenia eriensis dominated the community. The real question is, "Would both lakes have been the same had there been no cottages on Harp Lake?", and that is the question Lakeshore Capacity Study is trying to answer in the following manner.

The Lakeshore Capacity Study has chosen six Precambrian lakes in the Muskokas of mid-Ontario with varying amounts of development. Rather than express development as the number of cottages, P input to the lakes

Table 7: The retention coefficients of total phosphorus from septic tile filter beds of different characteristics. Results are based on four years of data (from Brandes et al. 1974).

Filter Bed	Rs
1. 22" Sand 8" mixture 4% red mud, 96% sand	0.76
2. 30" sand	0.34
3. 30" sand	0.22
4. 30" sand	0.48
5. 30" sand	0.01
6. 30" sand	0.04
7. 15" sand 15" sand 10% red mud, 90% sand	0.88
8. 15" sand 15" mixture, 50% limestone, 50% sand	0.73
9. 39" silty sand	0.63
10. 15" sand 15" mixture, 50% clayey silt, 50% sand	0.74

Table 8: Comparison of two lakes of similar morphometry, one with cottages and one without. (AHOD = areal hypolimnetic oxygen deficit, WOD = winter oxygen deficit).

		Harp	Jerry
Cottages		73	0
A_0	(km^2)	0.67	0.50
\bar{z}	(m)	12.4	12.4
V	(10^6m^3)	8.3	6.2
z_{\max}	(m)	36	35
A_d	(km^2)	4.0	7.3
[chl <u>a</u>]	($\text{mg} \cdot \text{m}^{-3}$)	2.8	2.6
S.D.	(m)	4.4	3.9
[P]	($\text{mg} \cdot \text{m}^{-3}$)	13	16
TON	($\text{mg} \cdot \text{l}^{-1}$)	0.24	0.30
AHOD	($\text{mg} \cdot \text{cm}^{-2} \cdot \text{d}^{-1}$)	0.034	0.034
WOD	($\text{mg} \cdot \text{cm}^{-2} \cdot \text{d}^{-1}$)	0.022	0.027
Algae		Cy.>Ba.*	Ba.>Cy.>Cry.*

* Cy = Cyanophyceae

Ba = Bacillariophyceae

Cry = Cryptophyceae

was estimated on the basis of the previous information. It has been assumed that the per capita P loading for the population is $0.8 \text{ kg}\cdot\text{yr}^{-1}$. The percent of the total input that could come from people is worked out and the progression of selected lakes appears according to Table 9. Natural P inputs to these lakes are presently being measured and assuming we can reliably predict what a lake's P concentration should be from natural loading, then deviations between measured values and the concentrations predicted on the basis of the natural inputs must be attributable to additional input. Or alternatively, by back-calculating the total loading from the lake's total mean P concentration and having measured the natural loading, the extra load can be determined by difference and this represents the P loading from anthropogenic sources.

In the preceding discussion it was necessary to convert load into a concentration (or vice versa) to provide the necessary translation of a lake with trophic status. To accomplish this requires using a P mass balance model. The choice of such a model can be made from an increasingly complex range of one box models to multi-compartmental models (Figs. 24, 25, 26). We have used a simple one-box or continuously-stirred tank reactor (CSTR) model which only considers input and output rates plus sedimentation of a substance, M, in a lake that is instantaneously and uniformly mixed (Rainey 1967, O'Connor and Mueller 1970, Vollenweider 1969, 1975). Other models increase the number of boxes by representing the layers of a stratified lake - epilimnion and hypolimnion and even metalimnion (Fig. 25) (Sweers 1969). These can be further complicated by adding other rate influencing processes such as production, decomposition, soluble and particulate fractions, etc. (Snodgrass and O'Melia 1975, Imboden 1974). Then, of course, there are the systems models which try to find solutions for coupled sets of differential equations representing the array of biological and chemical components, rather than modelling the behaviour of a single chemical species (Di Torro et al. 1975, Chen and Orlob 1975).

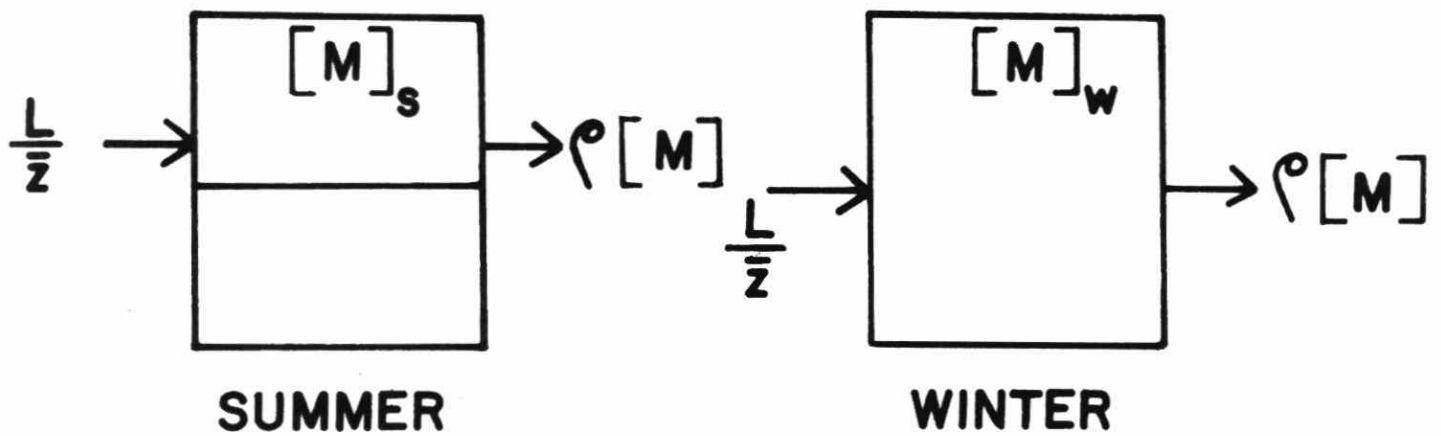
The Lakeshore Capacity Study model can be described by the equation in Fig. 27 where M_0 is the outflow concentration (Dillon and Rigler 1975). M_0 was originally equated with the average lake concentration but it can be approximated by the second expression (Chapra and Tarapchuk 1976). This empirically states that mean annual outflow total phosphorus, $[P]_0$ is 90%

Table 9: Estimated portion of total phosphorus supplied (J_T) by anthropogenic sources (J_A) to Lakeshore Capacity Study Lakes ($J = P$ supply in mg yr^{-1}).

Lake	$J_A/J_T(\%)$
Jerry	0
Red Chalk	4
Chub	26
Blue Chalk	27
Dickie	54
Harp	61

CLASS 2 - STRATIFICATION

2a) Conservative



eg. Sweers - pollutants in Great Lakes

2b) Non-conservative

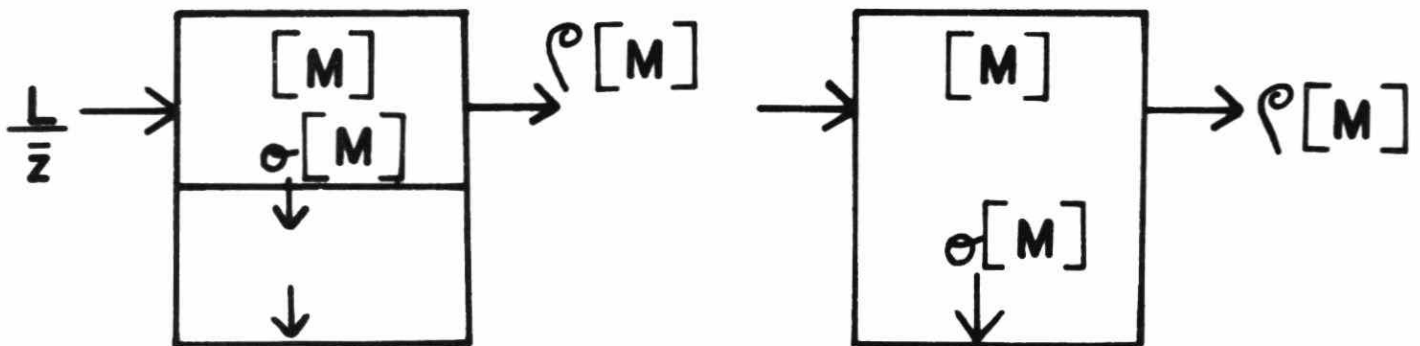
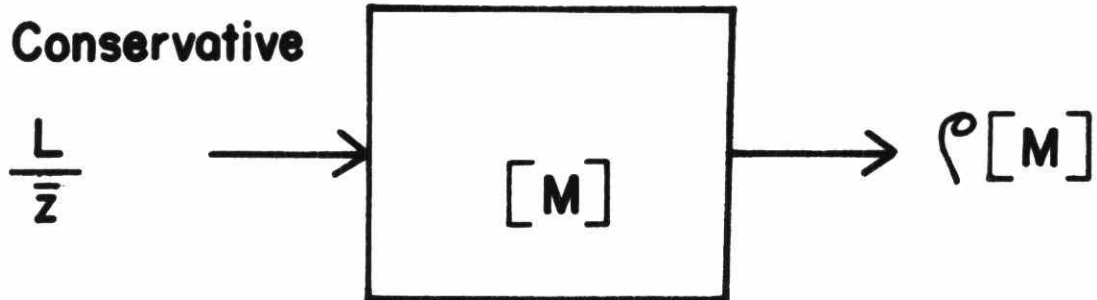


Fig. 25 Schematic representation of mass balance models that consider thermal stratification.

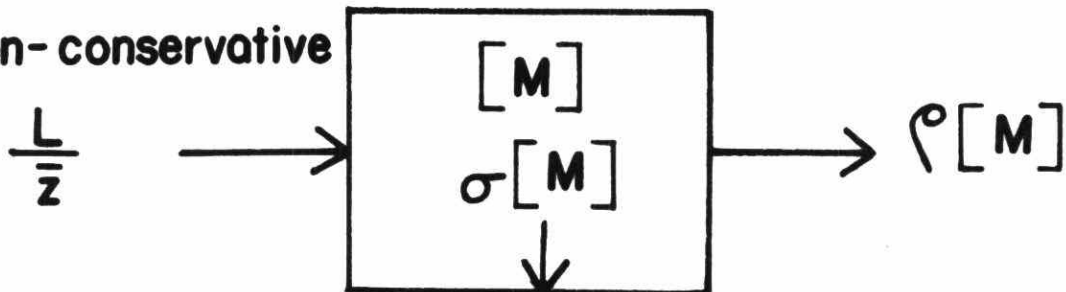
CLASS I ONE-BOX MODELS

Ia) Conservative



eg. Rainey - pollutants in Great Lakes
O'Connor & Mueller - Cl in Great Lakes

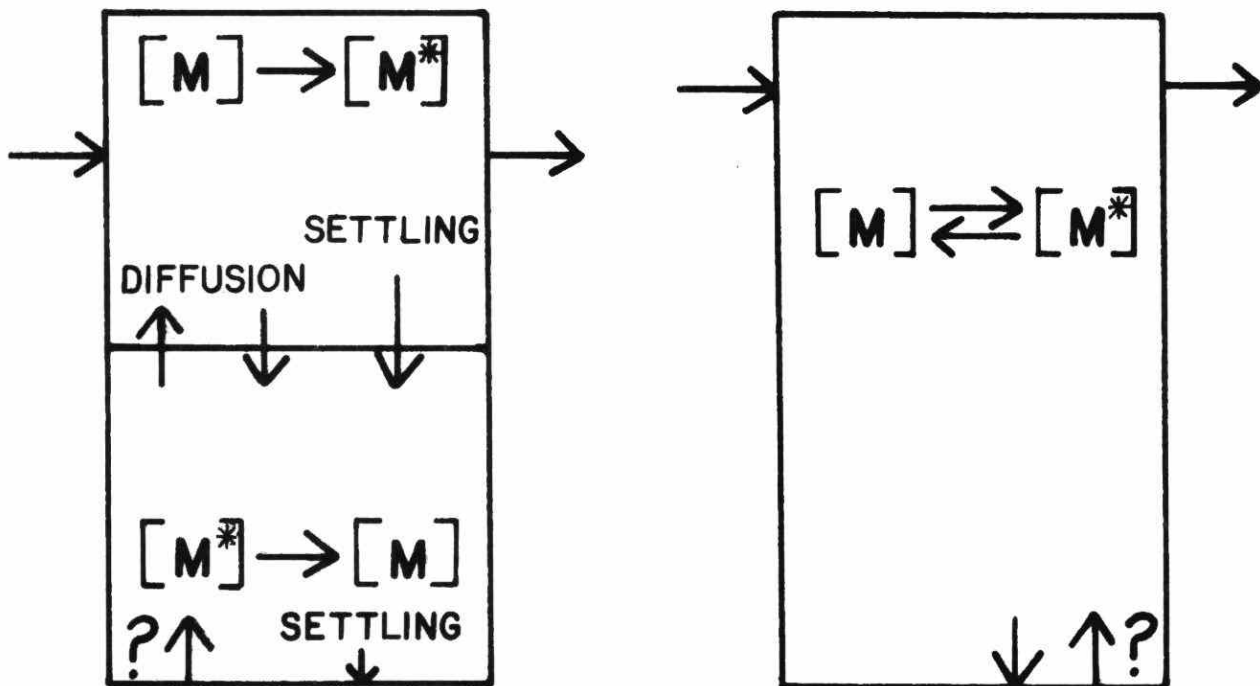
I b) Non-conservative



eg. Vollenweider - P in lakes

Fig. 24 Schematic representation of "one-box" mass balance models.

CLASS 3 - IN-LAKE EVENTS INCLUDED



eg. Snodgrass }
 Imboden } P in lakes

CLASS 4 - SPATIAL INHOMOGENEITIES & BIOLOGICAL COMPARTMENTS

eg. Thomann, Di Toro

Fig. 26 Schematic representation of mass balance models that include in-lake mechanisms.

$$\frac{d[M]}{dt} = \frac{L}{\bar{z}} - \sigma [M] - \frac{Q}{V} [M]_0$$

$$[P]_0 = 0.9 [P]$$

$$[P]_{ss} = L / \bar{z} (\sigma + 0.9 \rho)$$

or

$$[P]_{ss} = L(1-R) / 0.9 \bar{z} \rho$$

Fig. 27 Mass balance equation for a substance M, where L is the loading rate, \bar{z} the mean depth, σ the sedimentation rate, Q the outflow volume, V the lake volume, $\rho = Q/V$, $[M]_0$ the outflow concentration, R the retention coefficient, $[P]_{ss}$ the steady-state phosphorus concentration (where $M \equiv P$).

of the value of the mean lake total [P]. The steady-state can take on two forms, the second one containing the parameter R or the retention coefficient ($R = \text{the fraction of the total P input that does not leave a lake} = \sigma/(\sigma+p)$) (Dillon and Rigler 1974a).

The model was first tested by measuring all the required parameters (L, \bar{z}, ρ, R) for a series of lakes excluding the 0.9 factor (Dillon and Rigler 1974a). The actual measurements compared very well with the predicted P concentration at spring turnover (a mean deviation of $1.5 \mu\text{g l}^{-1}$) (Table 10), but when the 0.9 factor is included the predicted results are improved (a mean deviation of $1.0 \mu\text{g l}^{-1}$). Literature cases have also yielded excellent results using the steady-state equation.

For this or any other mass balance equation to be of general use, it should be possible to predict as many of the input variables as possible. Methodology for estimating P loading (L) was previously outlined and \bar{z} is usually obtainable for any lake using morphometric maps. This still leaves two parameters to estimate, ρ and R . Lake flushing rate or ρ can be approximated by deriving a water budget for a lake from precipitation, evaporation and evapotranspiration data. (Dillon and Rigler 1975). Such data have usually been isoplethed using long term averages for various geographic regions by government agencies and are readily available.

R has also been related to hydrologic parameters but with moderate success such as in Fig. 28 of R versus ρ times \bar{z} ($\rho \cdot \bar{z} = \text{hydraulic loading} = q \cdot s$) (Kirchner and Dillon 1975). Larsen and Mercier (1976) have tried other combinations of these parameters but none of these appears to have strengthened predictions of R . Before more empirical modelling is attempted in this area, some fundamental questions have to be answered, such as "Does R change as L changes?".

The preceding paragraphs have outlined a basic technique that allows prediction of the mean vernal P concentration in lakes. What remains to be done is to link this P concentration to factors indicative of trophic status.

One of the first relationships, studied by Dillon and Rigler (1974b) following Sakamoto's lead (1966), was the relationship between P concentration

Table 10: Comparison of measured phosphorus concentration at spring overturn with that predicted by a simple mass balance model (see Dillon and Rigler 1974a).

	[P] ($\text{mg}\cdot\text{m}^{-3}$)	$L(1-R)/\bar{z}_p$
Cameron	12.4	10.8
Four Mile	12.0	8.3
Bob	8.5	7.1
Boshkung	6.1	5.4
Halls	4.3	4.0
Beech	8.5	7.8
Maple	9.3	7.4
Pine	10.5	8.4
Cranberry	9.9	6.9
Moose	7.1	6.8
Haliburton	5.3	6.1

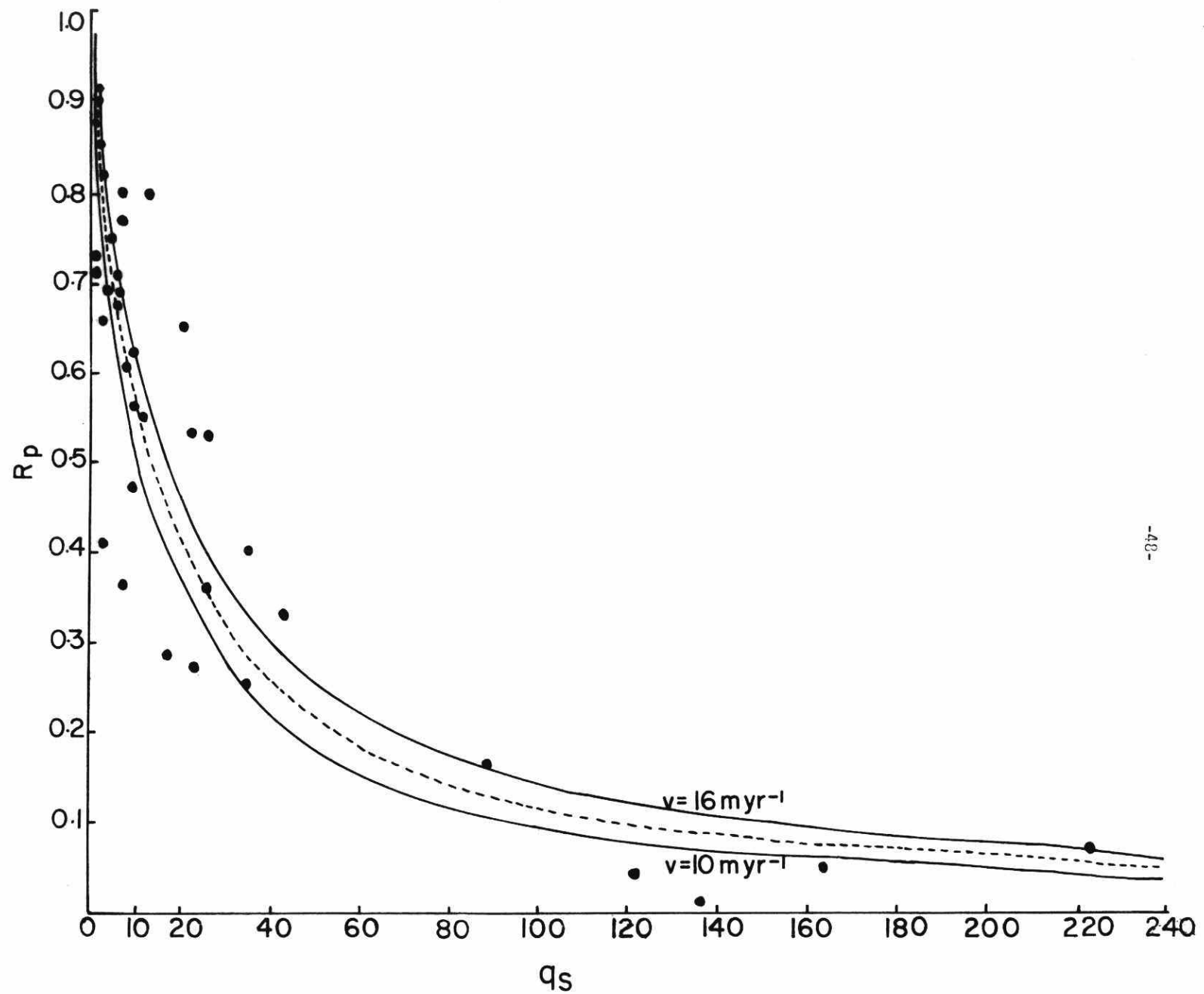


Fig. 28. Relationship between phosphorus retention coefficient (R_p) and hydraulic load ($q_s = \bar{z}_p$). The settling velocity v can be shown to be equal to \bar{z}_p .

at spring turnover and mean summer chlorophyll a. Fig. 29 is the original data set used, including Sakamoto's data (Dillon and Rigler 1974b), while Fig. 30 shows the addition of later points. Although the relationship appears to work very well, it is a log-log relationship, so it has limited prediction capabilities at both of its tail-ends.

Because of the relative ease and effort required to measure chlorophyll a, many P-chlorophyll a plots have been published in the literature (Fig. 31) and great differences between one another have been shown. Nicholls and Dillon (1978) have attempted to analyze the reasons for such differences (see Appendix A for further discussion).

Secchi disc depths have been estimated from chlorophyll a (Fig. 32) by a number of people. There are three factors affecting light extinction and thus the correlation between Secchi disc and chlorophyll a: 1) particulate material containing plant pigments, 2) particulate material not containing plant pigments and 3) dissolved colour. The second factor may be of little importance in Precambrian lakes, but colour frequently is. Colour can be factored out quite easily by multiple regression analysis.

Relationships between average total phosphorus concentration and phytoplankton biomass (cell volume) and between phytoplankton biomass and Secchi disc visibility are generally better defined than those substituting chlorophyll as an indicator of phytoplankton biomass. However, chlorophyll will surely remain a popular measure of algal biomass due to the economics and time constraints of most research programmes.

Ideally, one would like to measure parameters such as P loading and concentration and predict the genera or species of zooplankton, phytoplankton and macrophytes (etc.) to be found in a lake. Nicholls (1977) found that the greater the algal biomass in a lake, the less important the Chrysophyceae are as contributors to this biomass (Fig. 33). When this is coupled with P data, it is apparent that the Chrysophyceae disappear from a lake ecosystem at a P concentration greater than $30 \mu\text{g l}^{-1}$ (Fig. 34) for reasons explained earlier. This kind of information is of particular value to water managers. For example, the small motile Chrysophyceae are an important nutritive source for the zooplankton. In fact, their small size and hence rapid turnover suggests their importance in the production ecology of a lake is probably much greater than measurements of their biomass would indicate. The loss of this group from a eutrophying lake clearly has important

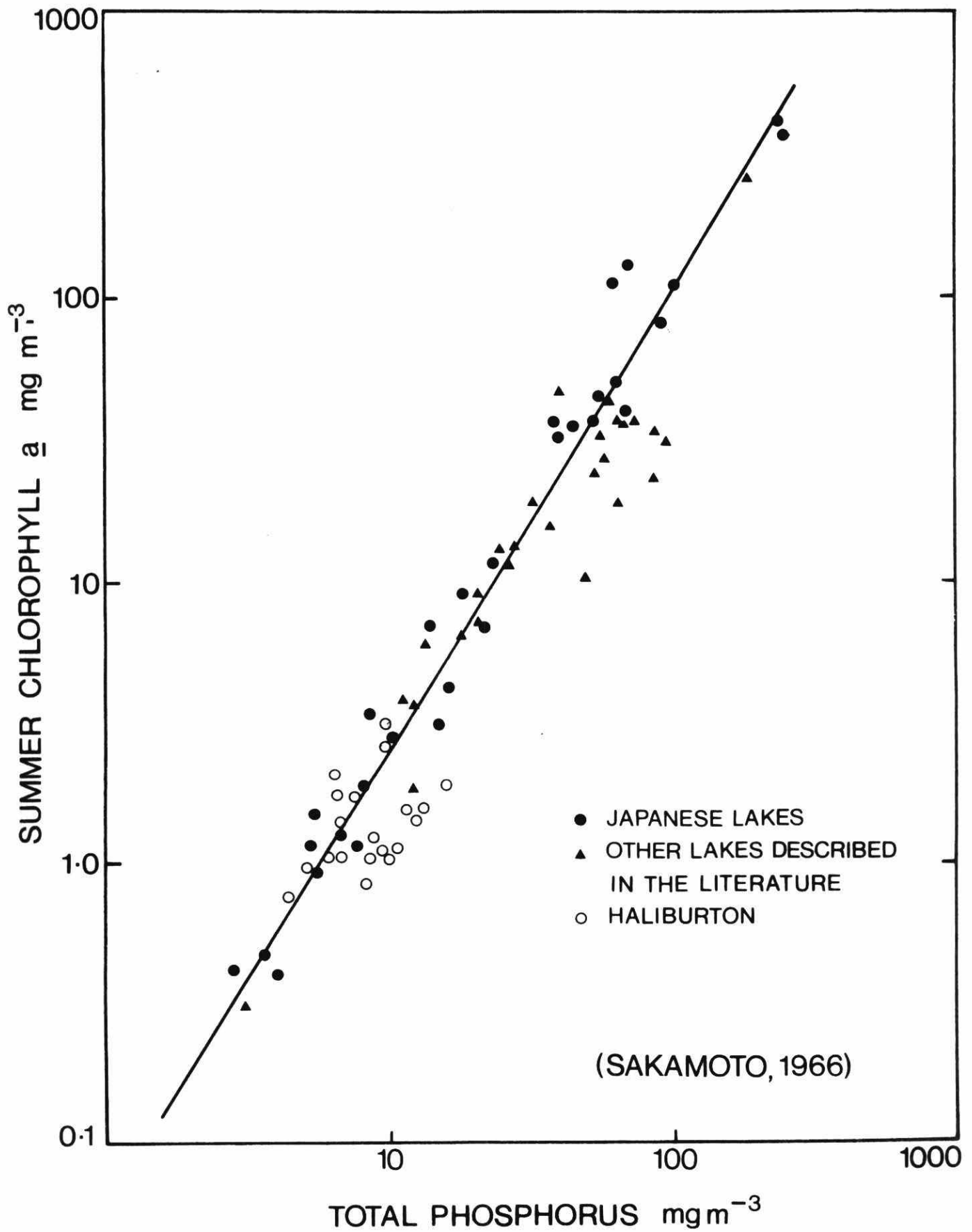


Fig. 29 Relationship between total phosphorus concentration at spring overturn and average summer chlorophyll \bar{a} concentration (from Dillon and Rigler 1974b).

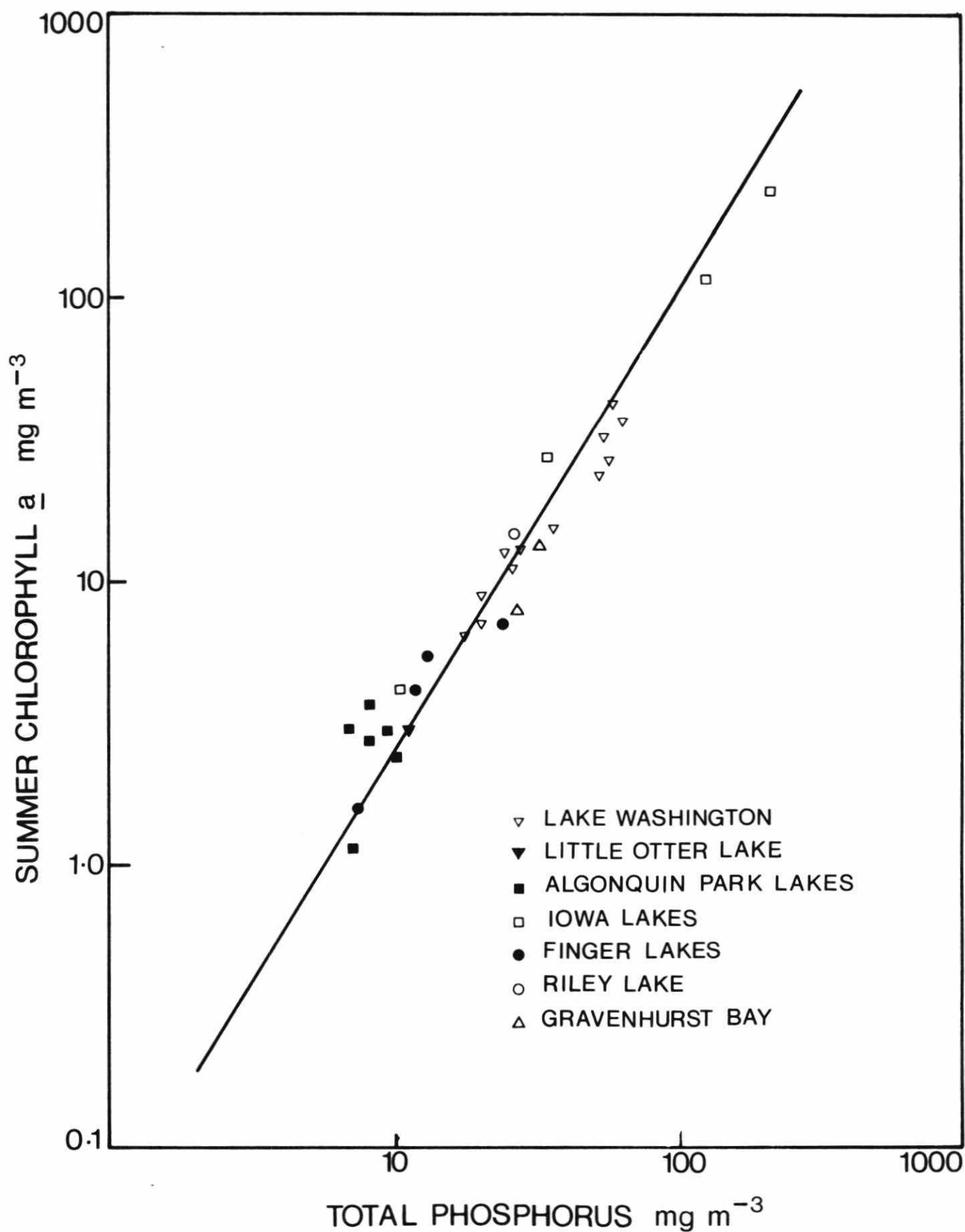


Fig. 30 Additional data shown with the original line generated in Fig. 29 (Dillon and Rigler 1975b).

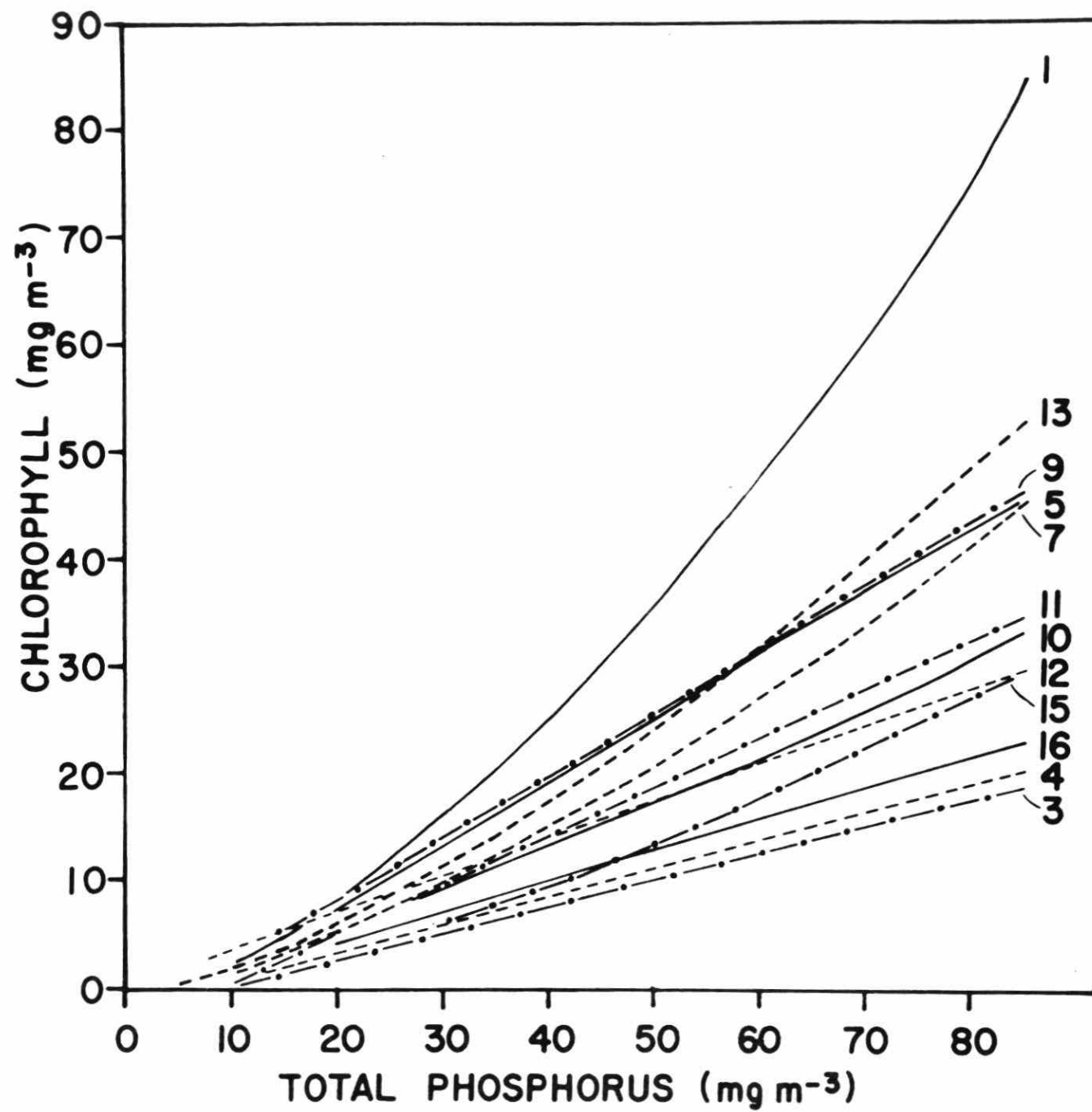


Fig. 31. Phosphorus - chlorophyll curves reported in the literature (Nicholls and Dillon 1978).

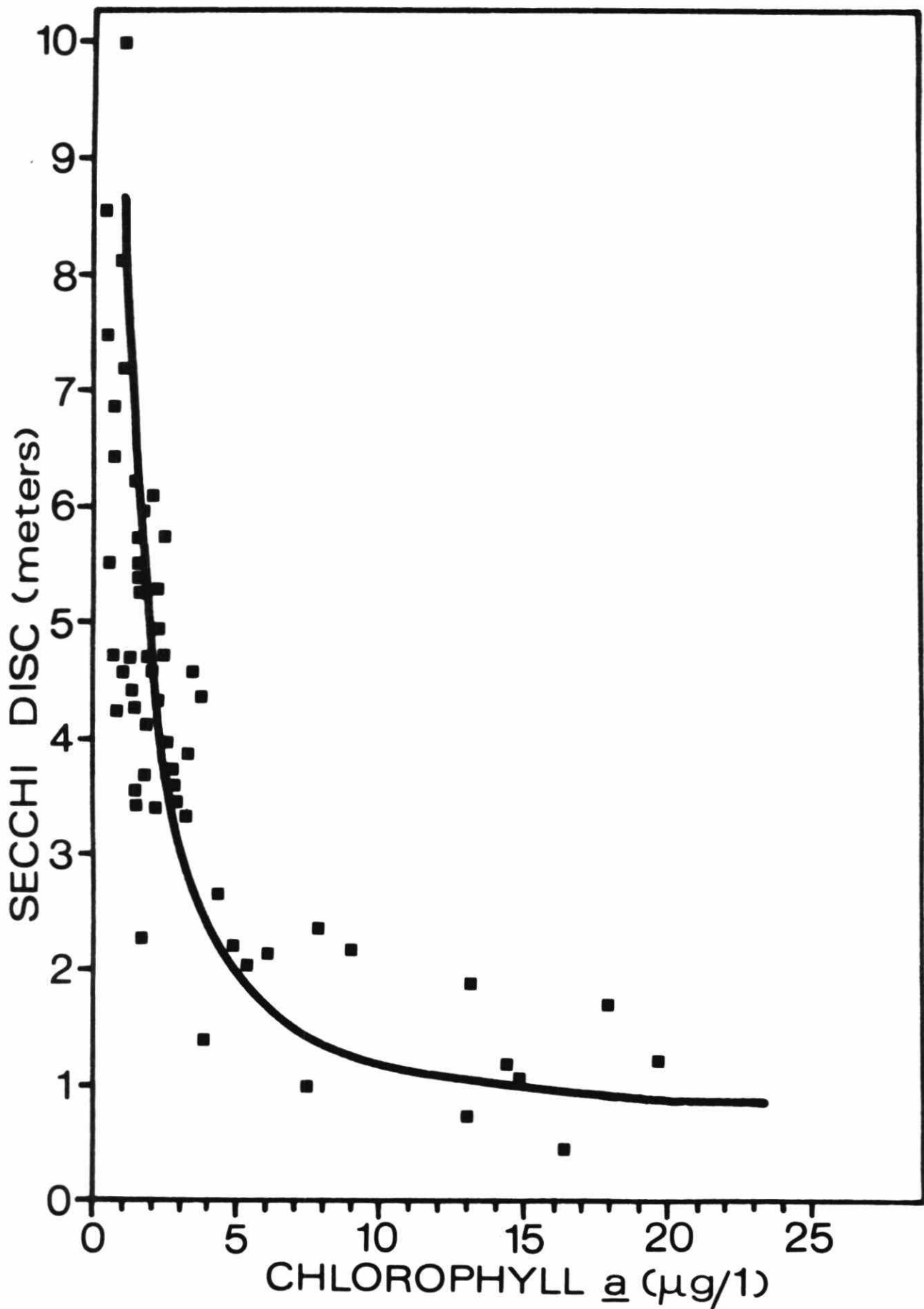
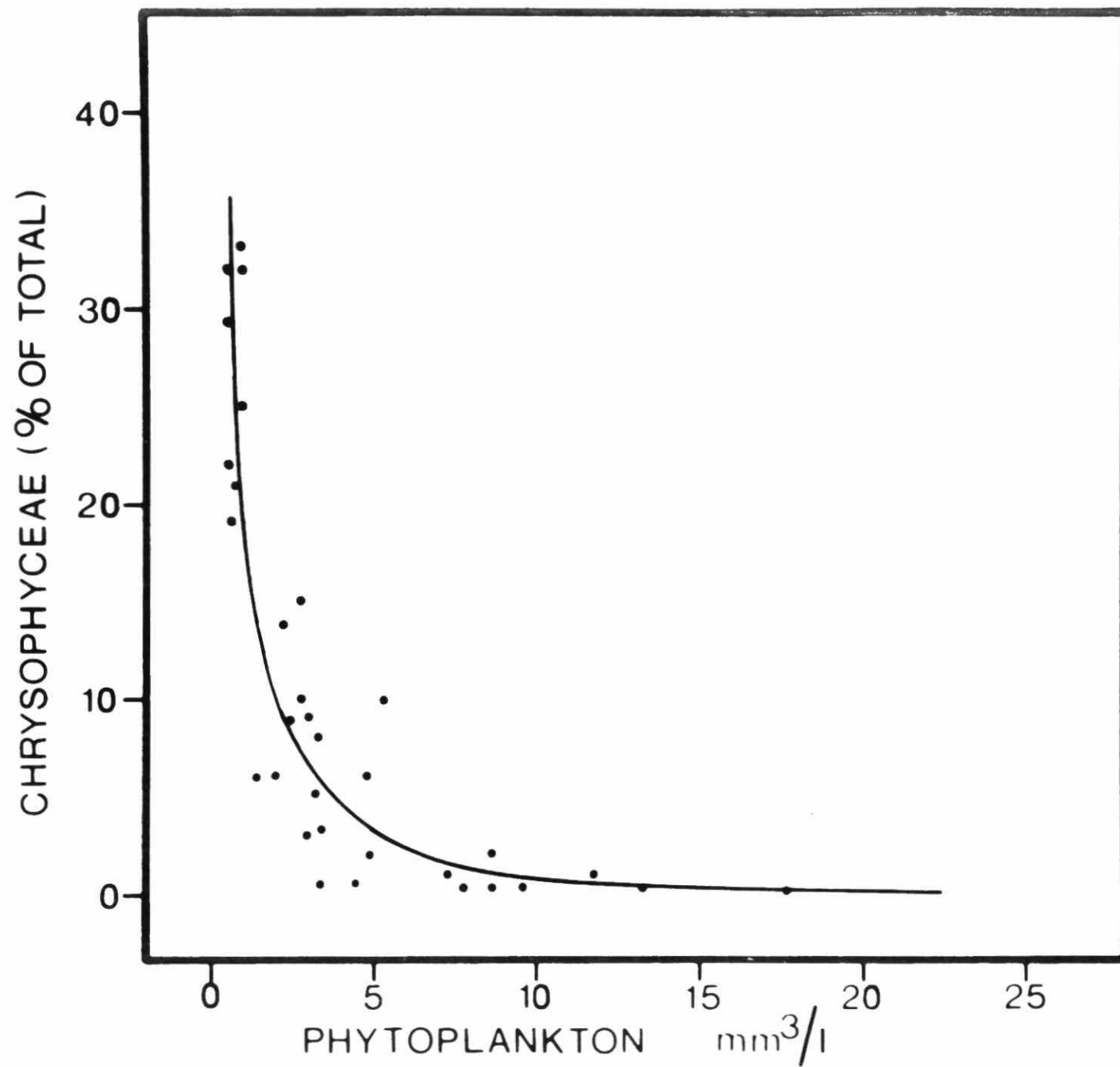


Fig. 32. Relationship between Secchi disc depth and chlorophyll a concentration (summer average) in the euphotic zone of Ontario lakes. Data are taken from the Ontario Ministry of the Environment recreational lakes programme.



54

Fig. 33. Relationship between total phytoplankton biomass and percentage of the biomass contributed by Chrysophyceae (from Nicholls 1977).

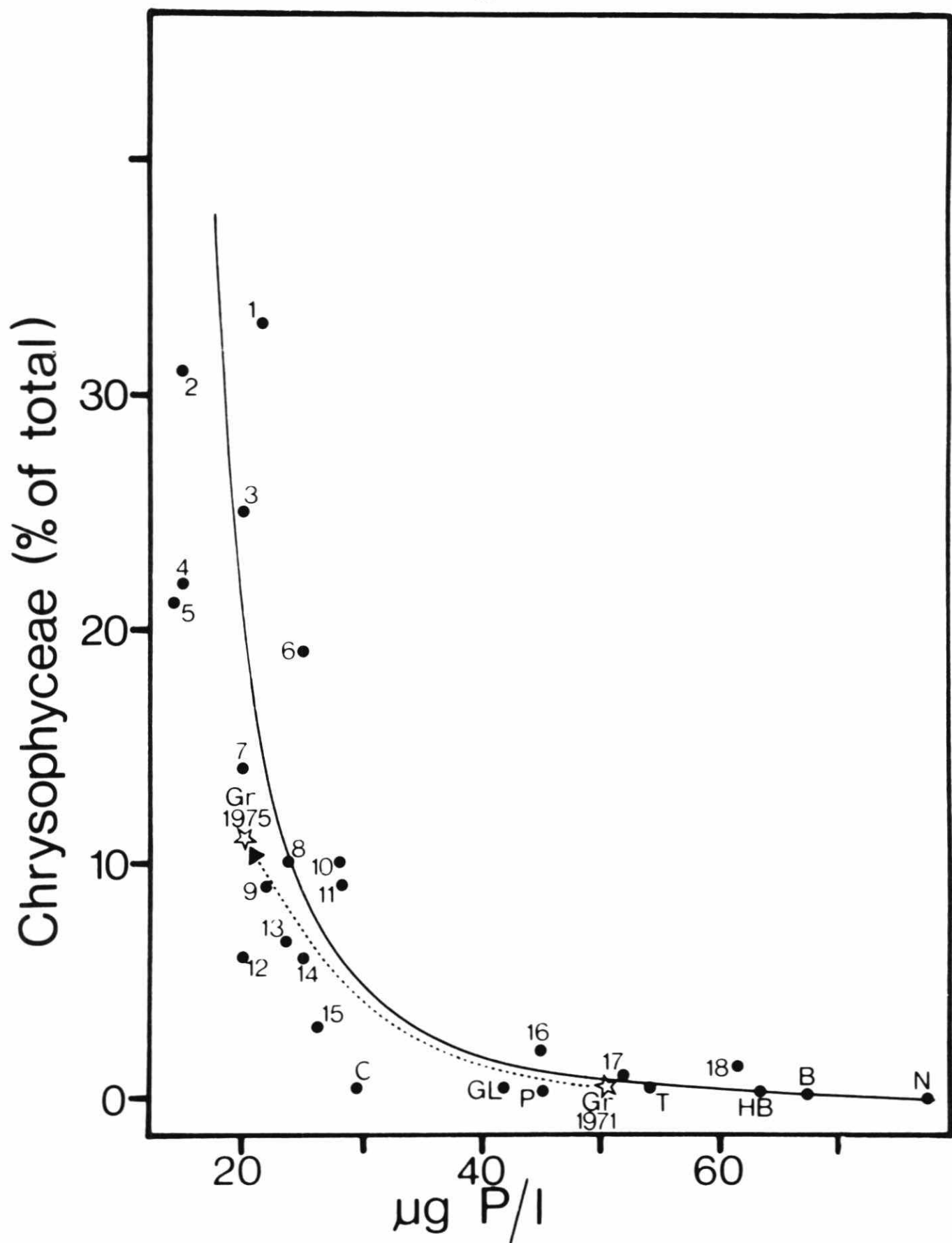


Fig. 34. Relationship between total phosphorus content of lakes and the proportion of the algal biomass contributed by the Chrysophyceae.

implications on the food chain and ultimately, the fisheries. Consequently, Nicholls (1977) has developed a nomogram (which also considers the relative importance of the Chrysophyceae, expressed as a percentage of the total phosphorus) to interrelate these indices of trophic status for predictive purposes (Fig. 35). An example of the use of these techniques is outlined. The P budget for Gravenhurst Bay was estimated before and after P removal (Table 11). The anthropogenic load from the sewage treatment plant was measured while the other figures were estimated or extrapolated. The 1975 predicted and measured values for several "trophic status indices" (Table 12) suggest that changes in P concentration can yield quantitatively predictable results.

The inherent assumption in the use of a phosphorus-phytoplankton relationship for predictive purposes is that the total phosphorus measurement includes all phosphorus in solution and suspension which, if not already incorporated into algal cells, is potentially available. Clearly there may be some forms of phosphorus in lake water which are not available, yet it does not seem practical to determine routinely the "biological P" in lake water since such studies involve rather sophisticated bioassay techniques. It is nevertheless important to recognize in nutrient budget studies that a portion of the phosphorus load to a lake may never be available for plant growth in the lake (for example see our results of algal assays on Lake Timiskaming inflowing streams) (Fig. 36).

Although there is now much data to provide strong evidence that algal growth is controlled by phosphorus concentration in most lakes, especially those of the Precambrian Shield, there is also evidence that very productive waters may be nitrogen controlled (e.g. Nicholls 1976a). Additionally, we have completed laboratory algal assays on the inner Bay of Quinte and Thompson Lake and have demonstrated nitrogen limitation in these highly eutrophic waters (Fig. 37, 38 and 39).

These findings suggest important implications for the recovery of the Bay of Quinte following implementation of controls on phosphorus loading from municipalities within the Quinte basin. Because the inner bay photoplankton is nitrogen limited and the outer bay phytoplankton, phosphorus limited, the response to decreased phosphorus loading from sewage treatment

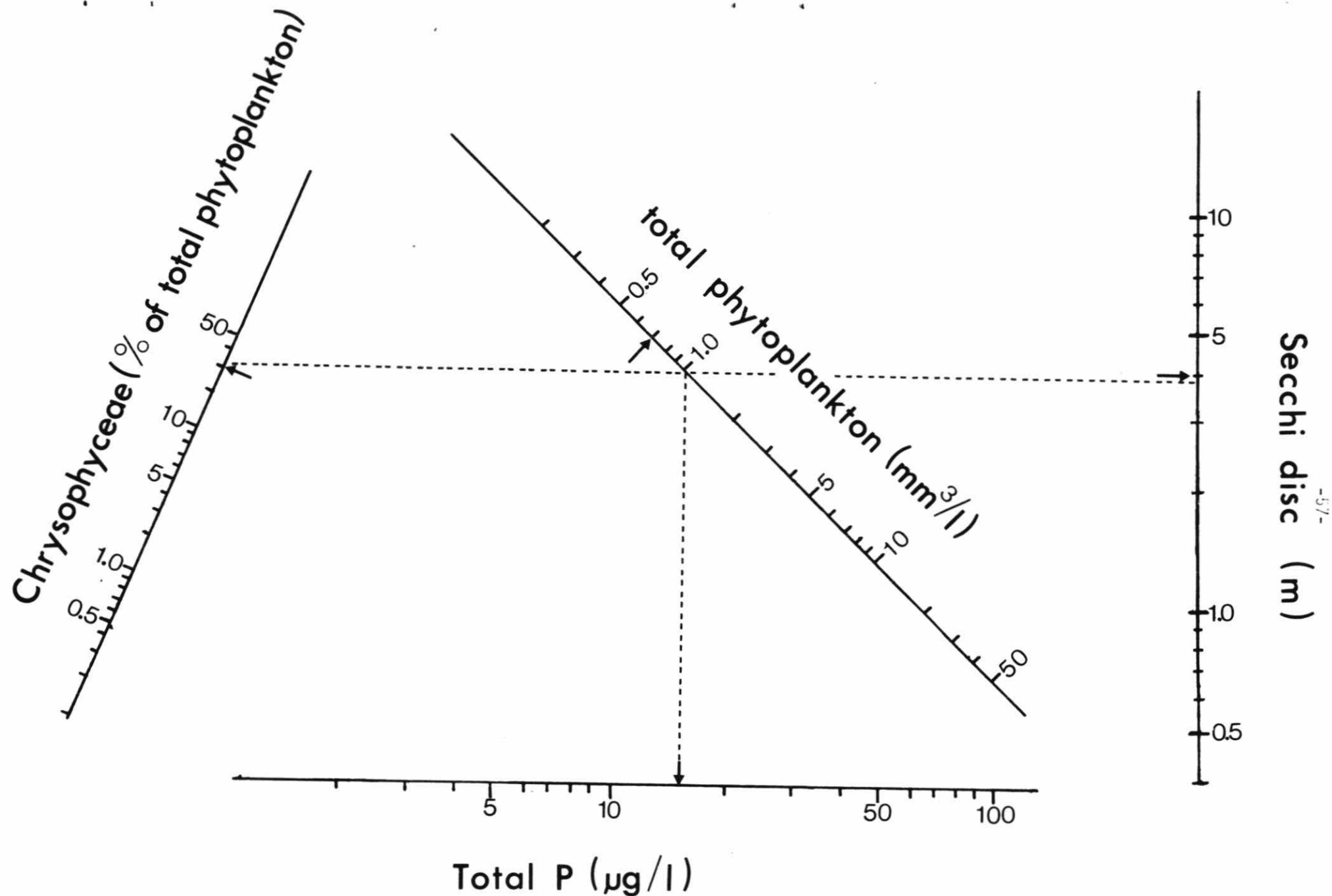


Fig. 35 Nomogram facilitating intercomparisons of the most reliable trophic status indicators. To illustrate the use of the nomogram, for predictive purposes, Balsam Lake average total phosphorus concentration and the resulting predicted values of total phytoplankton biomass, Chrysophyceae abundance and Secchi disc visibility are indicated. The arrows indicate the measured values.

Table 11: Estimated Phosphorus budget for Gravenhurst Bay (from Dillon et al. 1978).

Source	1971	1975
Land drainage (kg)	180	180
Cottages (kg)	520-1290	280-690
Municipal (kg)	2580	680
Precipitation (kg)	150-310	150-310
Total (kg)	3430-4360	1280-1860
Load ($\text{g}\cdot\text{m}^{-2}\cdot\text{yr}^{-1}$)	0.82-1.06	0.31-0.45

Table 12: Comparison of predicted and measured trophic status parameters for Gravenhurst Bay following P removal.

		Measured	Predicted
$[P]_{SP}$	$(mg \cdot m^{-3})$	26	20-29
$[P]_{SS}$	$(mg \cdot m^{-3})$	20	17-25
$[chl\ a]$	$(mg \cdot m^{-3})$	5.0	5.6
S.D.	(m)	3.9	2.7 (3.0 from nomogram)
AHOD	$(mg \cdot cm^{-2} \cdot d^{-1})$	0.025	0.035
Phytoplankton biomass	$(mm^3 \cdot l^{-1})$	2.5	2.0
Chrysophyceae (% of total)		11	15

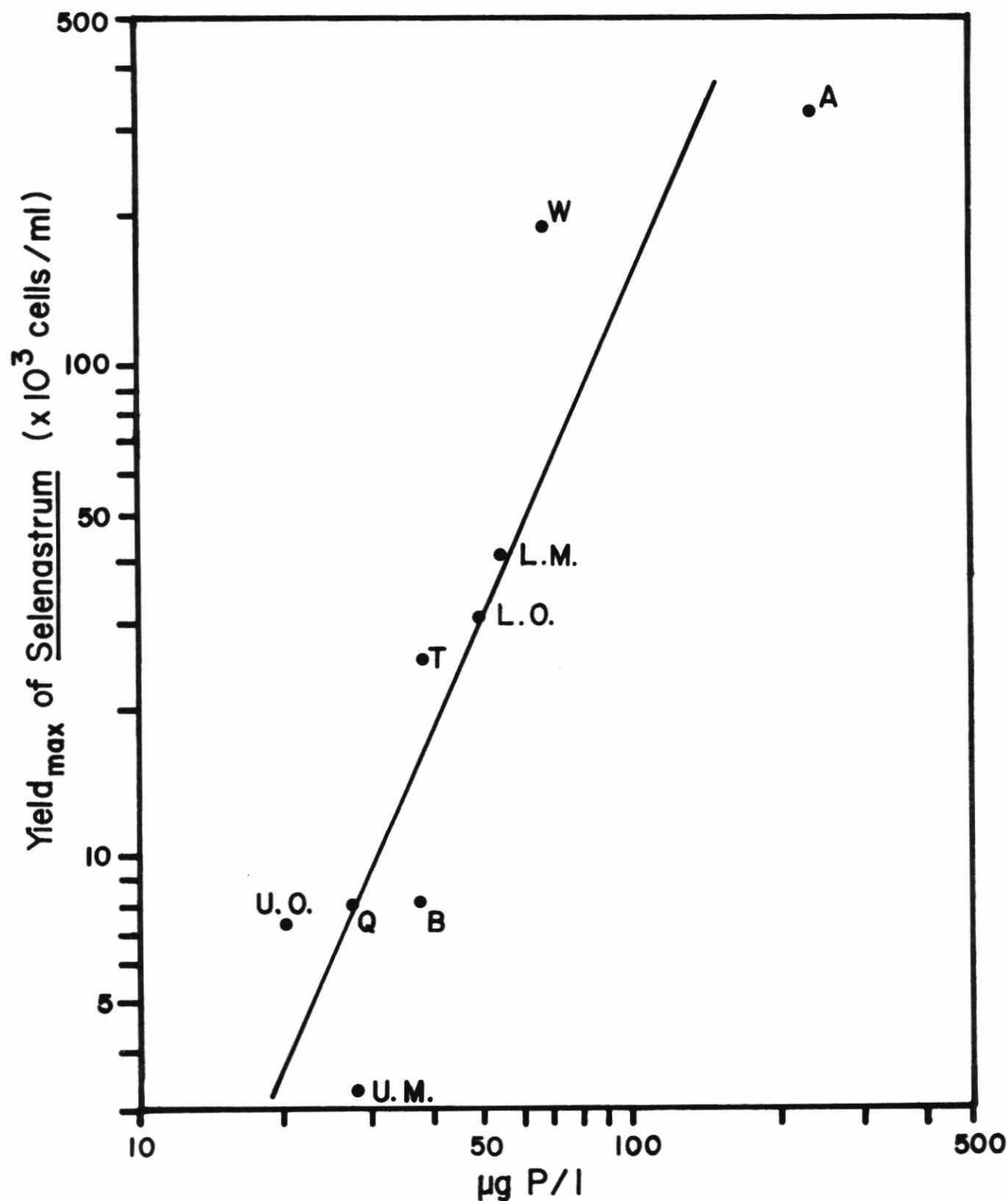


Fig. 36. Laboratory algal assay of stream water inputs to Lake Timiskaming. Keyed as follows: A, Alligator Creek; W, Waki Creek; L.M., Lower Moffat Creek; L.O., Lower Otterskin Creek; B, Blanch River; Q, Quince River; U.M., Upper Moffat Creek; U.O., Upper Otterskin Creek; T, Lake Timiskaming (mid-lake). To some extent, the poor fit of the data suggest that a portion of the total phosphorus measured at some stream locations is not available for algal growth.

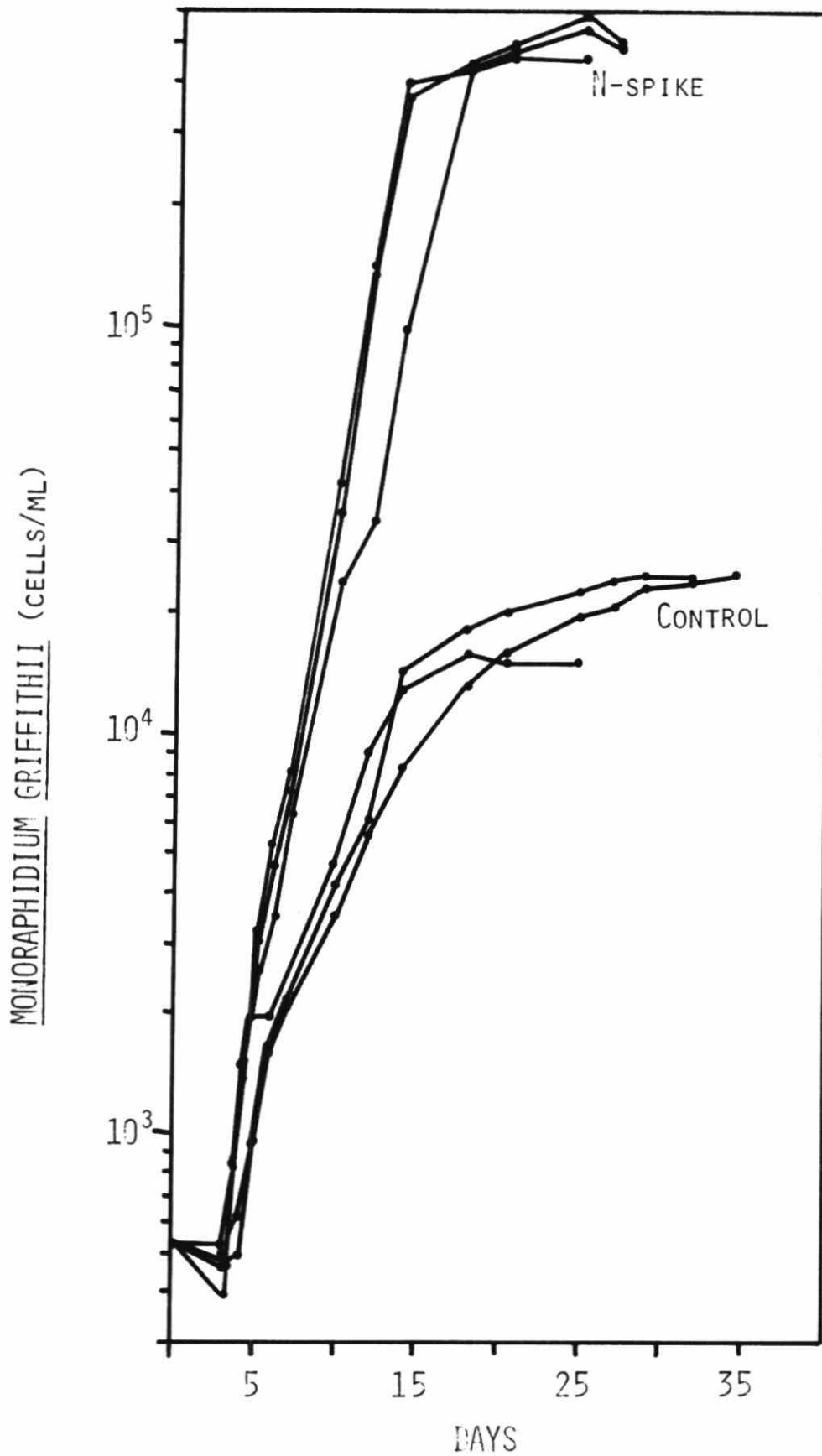


Fig. 37. Growth of Monoraphidium griffithii in filtered water from the inner Bay of Quinte (control) and showing the increased yield of the water with added NaNO_3 (1,125 mg N/l). Three additional samples spiked with KH_2PO_4 (50 μg P/l) showed no significantly increased yield over the control.

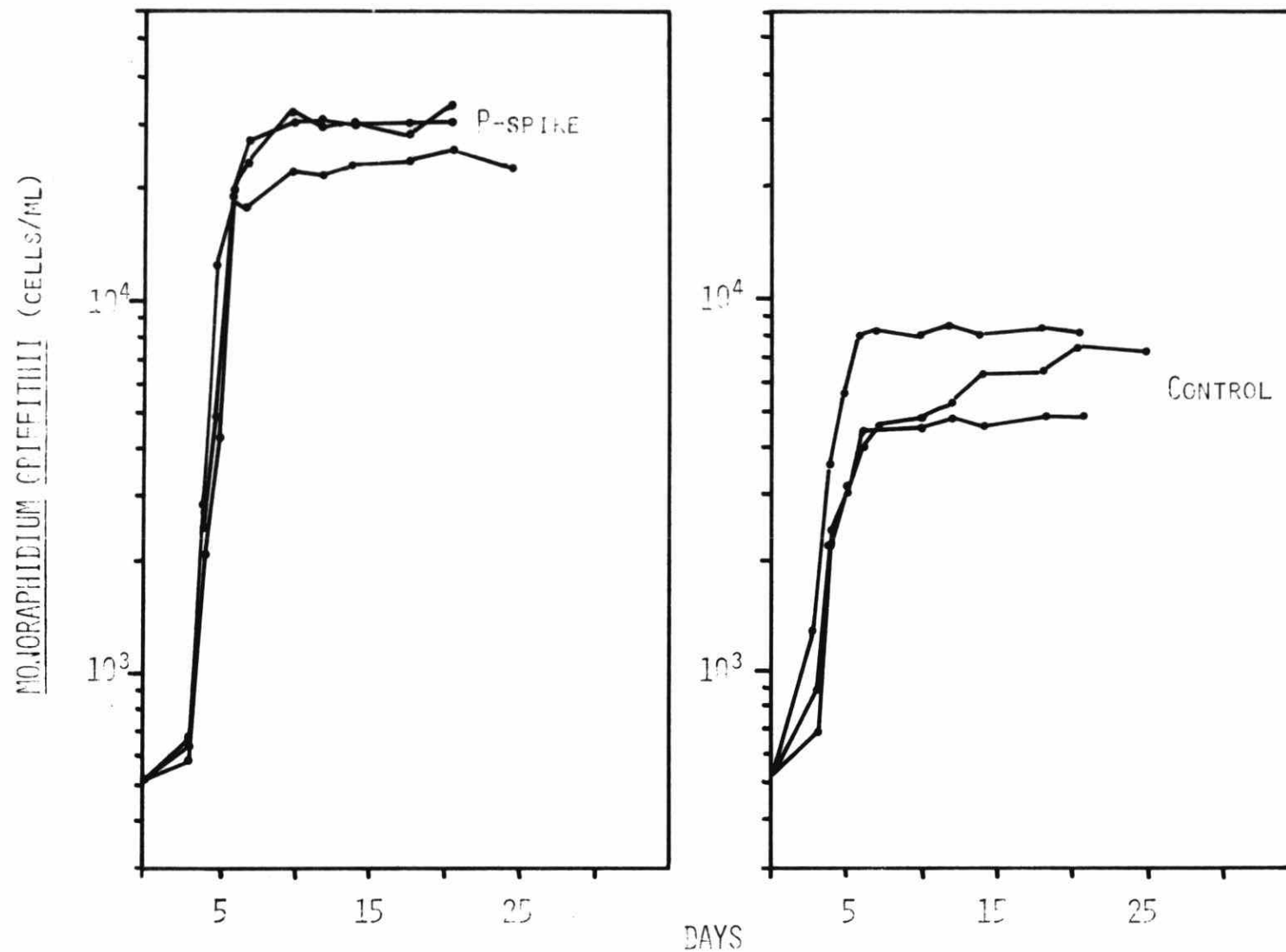


Fig. 38. Growth of *Monoraphidium griffithii* in filtered water from the outer Bay of Quinte (control) and showing the increased yield of the water with added KH_2PO_4 ($50 \mu\text{g p/l}$). Three additional samples spiked with NaNO_3 ($1,125 \text{ mg N/l}$) yielded lower maximum biomass than either the control or the P-spiked samples.

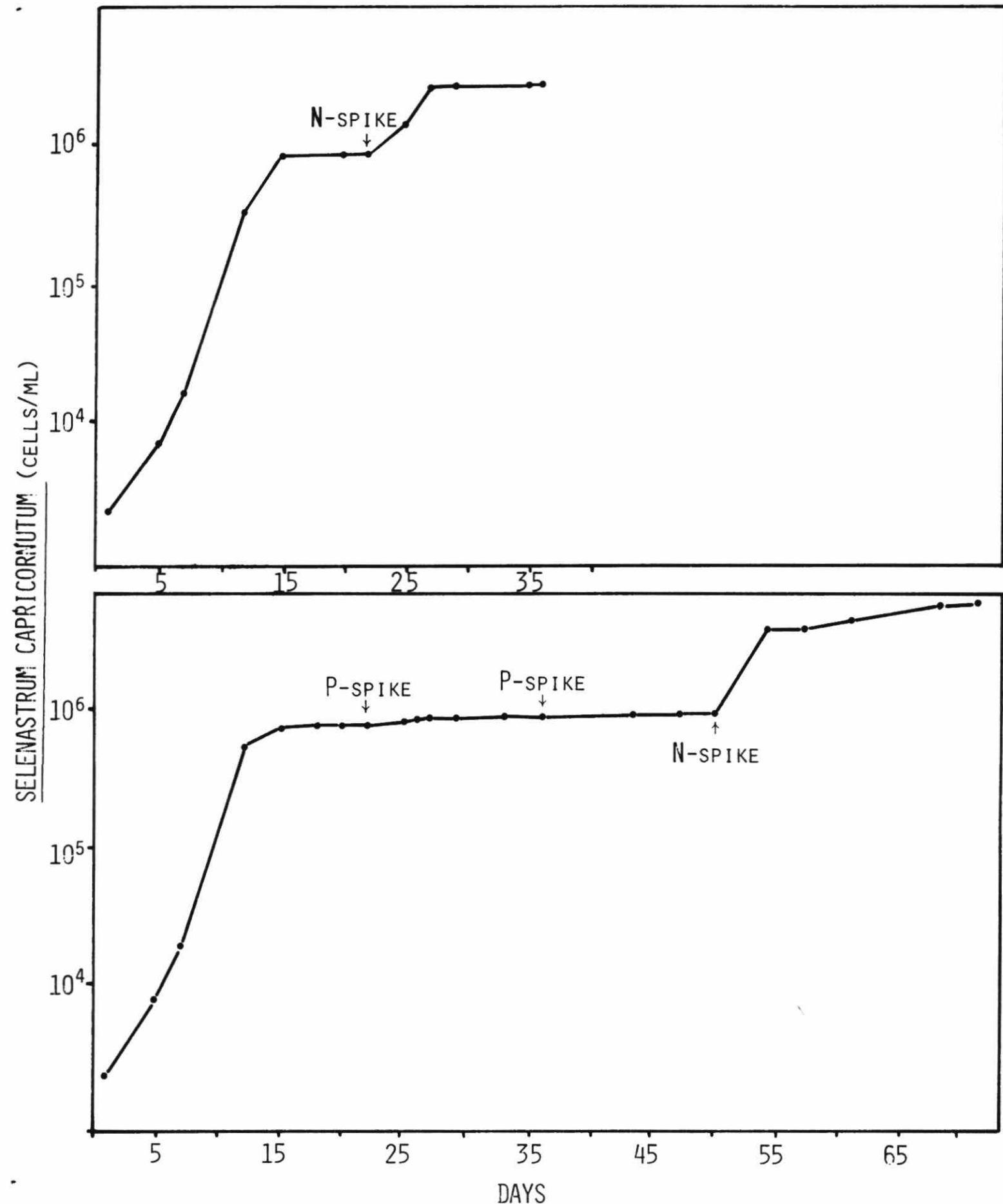


Fig. 39 Growth of *Selenastrum capricornutum* in filtered water from Thompson Lake showing growth response to additions of inorganic phosphorus and nitrogen. A similar experiment showed no response to additions of iron and manganese.

plants should be noticed first at the outer bay locations. In the inner Bay of Quinte, phosphorus will have to be established as the controlling nutrient before further reductions in concentration permit declines in algal biomass.

Nitrogen limitation of phytoplankton in highly eutrophic lakes generally does not curtail the obnoxious bloom-forming species (e.g. Aphanizomenon and Anabaena spp.) which can fix gaseous nitrogen. In fact, in Thompson Lake (15 km N of Toronto), these bloom-forming species were dominant during most of the summer period prior to attempts to control them by artificially destratifying the lake (Fig. 40). By means of their gas vesicles, blue-green algae are unique in their ability to alter their buoyancy and select depths which are optimal for their growth. A stable water column is therefore essential for the effective use of their buoyancy regulating mechanism. That certain blue-green algae can be essentially eliminated from a thermally stratified lake by mixing the water has been known for some time (e.g. Sirenko et al. 1972, Karasik and Stovbun 1974).

When a eutrophic lake is artificially mixed (for example, by aeration), blue-green algal cells are forced to spend a disproportionate amount of time in the deeper, "aphotic" water which promotes excessive formation of gas vesicles within the cell. The buoyant cells are then more prone to surface floatation where they are killed by sunlight (ultraviolet exposure and/or photo-oxidation of pigments). During 1974, when Thompson Lake ($A_0 = 3.9$ ha, $Z_{\max} = 26$ m) was being adequately mixed by the aerator, blue-green algae were almost non-existent in the lake (Fig. 41). Instead the lake was populated by small coccoid green algae, cryptomonads and diatoms, forms which fit well into aquatic food chains and promote development of larger fish-food organisms. Owing mainly to their large colonial habit and the presence of a protective mucillagenous sheath, most blue-green algae are not an important component of the food chain.

It is worth emphasizing that lake systems are rarely so simplified that a single stress illicit a single response, and in some eutrophic lakes, lake response to artificial mixing apparently can be much more complex. Heart Lake ($A_0 = 14.3$ ha, $Z_{\max} = 10.4$ m), which has also had a

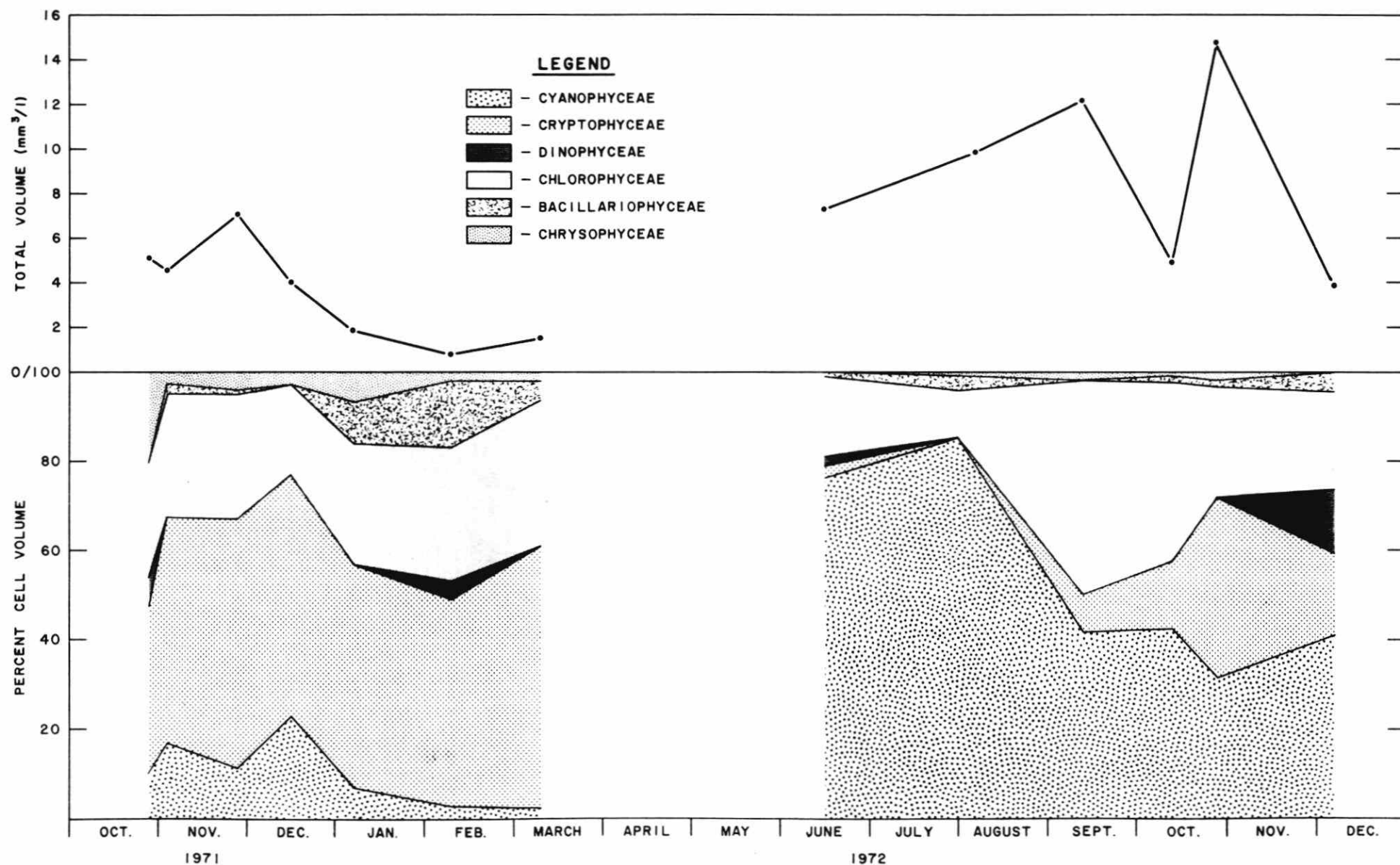


Fig. 40. Phytoplankton of Thompson Lake during 1972. Note the high biomass of blue-green algae (Cyanophyceae).

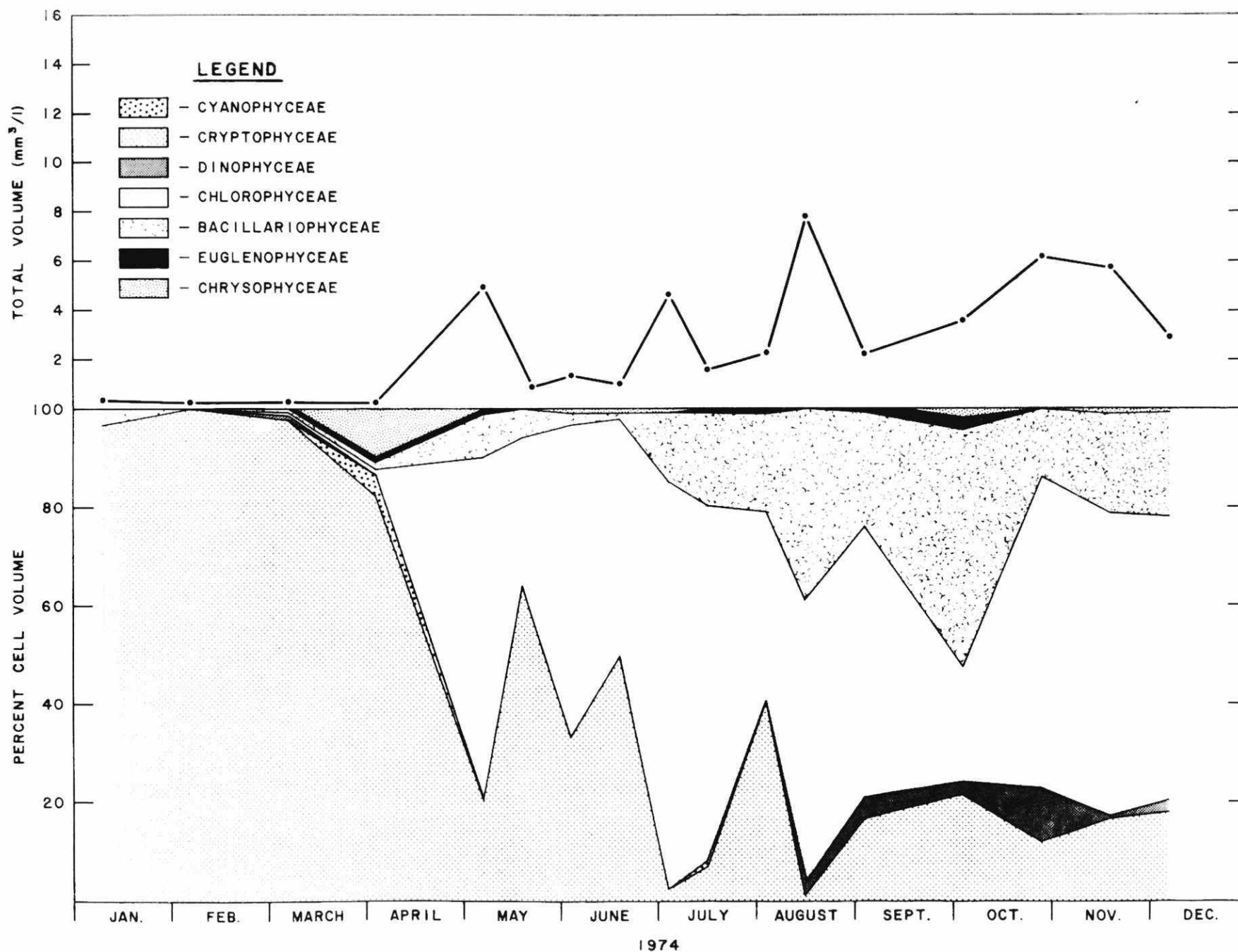


Fig. 41. Phytoplankton of Thompson Lake during 1974. Note the virtual absence of blue-green algae (Cyanophyceae).

history of blue-green algal problems, has been aerated since 1975. Artificial destratification again apparently resulted initially in replacement of blue-green algae by "food-chain functional" forms such as small greens and cryptomonads. However, an additional effect of aeration was to increase the habitat of Daphnia pulex by replenishing the oxygen content of the deeper waters (Strus 1976). Because the D. pulex could now take refuge in the dark, deeper, oxygenated water from planktivorous fish, they were therefore able to thrive under the conditions created by aeration. The development of the Daphnia population was so successful that the desirable forms of algae were consumed at a rate almost equal to their production. By this stage in the sequence of events, the large dinoflagellate Ceratium hirundinella had appeared (as it usually does in July and August in eutrophic lakes). It is too large to be grazed by Daphnia pulex, and in the absence of other algae, C. hirundinella experienced no competition for light and nutrients and a massive population developed (Fig. 42). The sudden death of this massive dinoflagellate population resulted in a summerkill of the lake.

Clearly, some lake management techniques have complex ramifications and a thorough understanding of lake processes in a lake-by-lake assessment would seem to be a necessary requisite to any successful management programme.

In summary, the points that we have tried to make are the following:

- 1) That P is the element controlling trophic status in most lakes and that biological changes can be induced by changing P concentration by even very small amounts. Under some highly artificial conditions, N or C may control algal growth. N limitation especially can be easily detected with a simple laboratory algal assay.
- 2) That the input of P to a lake can be estimated without a lot of collection of raw data.
- 3) That lake morphometry, hydrology and phosphorus load can be combined in a mass balance model to give a good prediction of phosphorus concentration.

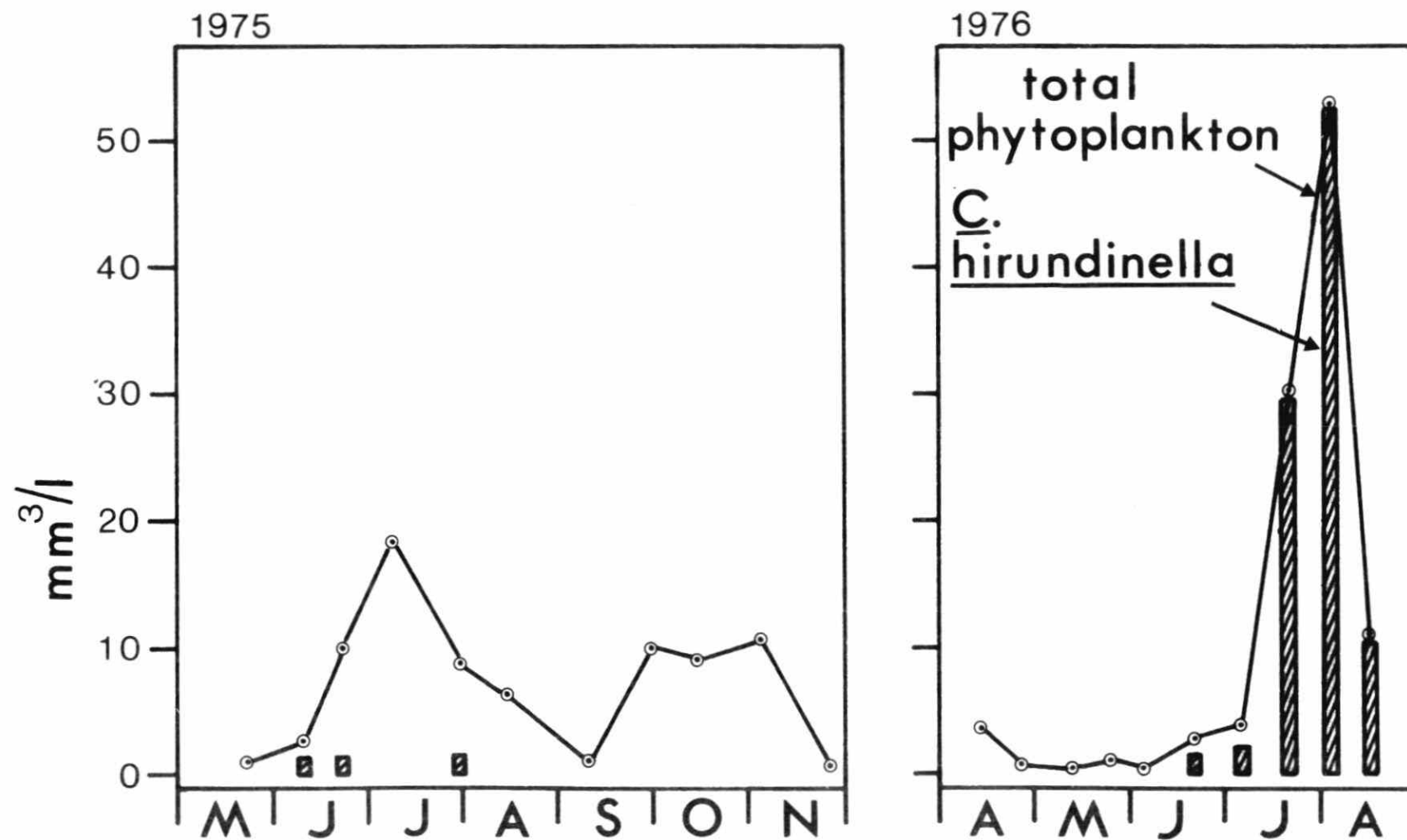


Fig. 42. Seasonal distribution of total phytoplankton and of *Ceratium hirundinella* in Heart Lake during the first and second years of artificial destratification.

- 4) That P can be used to let us estimate phytoplankton biomass and some characteristics of the composition which have important implications for lake management (e.g. hypolimnetic dissolved oxygen depletion rates, efficiency within the food web of the lake, water clarity, etc.).

REFERENCES

- BRANDES, M., N.A. CHOWDRY and W.W. CHENG. 1974. Experimental study on removal of pollutants from domestic sewage by underdrained soil filters. National Home Sewage Disposal Symposium, Am. Soc. Agric. Eng., Chicago, Ill., 1974.
- CHAPRA, S.C. and S.J. TARAPCHAK. 1976. A chlorophyll *a* model and its relationship to phosphorus loading plots for lakes. *Water Resour. Res.* 12:1260-1264.
- CHEN, C.W. and G.T. ORLOB. 1975. Ecological simulation for aquatic environments. p.475-588. In B.C. Patten (ed.). *Systems Analysis and Simulation in Ecology*. Vol. III. Academic Press, N.Y.
- DAVIS, C.C. 1964. Evidence for the eutrophication of Lake Erie from phytoplankton records. *Limnol. Oceanogr.* 9:275-283.
- DILLON, P.J. 1975. The phosphorus budget of Cameron Lake, Ontario : the importance of flushing rate to the degree of eutrophy of lakes. *Limnol. Oceanogr.* 20:28-39.
- DILLON, P.J. and W.B. KIRCHNER. 1975. The effects of geology and land use on the export of phosphorus from watersheds. *Water Res.* 9:135-148.
- DILLON, P.J., K.H. NICHOLLS and G.W. ROBINSON. 1978. Phosphorus removal at Gravenhurst Bay, Ontario : an 8 year study on water quality changes. *Verh. Internat. Verein. Limnol.* 20:in press.
- DILLON, P.J. and F.H. RIGLER. 1974a. A test of a simple nutrient budget model predicting the phosphorus concentration in lake water. *J. Fish. Res. Bd. Canada* 31:1771-1778.
- _____ 1974b. The phosphorus-chlorophyll relationship in lakes. *Limnol. Oceanogr.* 19:767-773.
- _____ 1975. A simple method for predicting the capacity of a lake for development based on lake trophic status. *J. Fish. Res. Bd. Canada* 32:1519-1531.
- DI TORO, D.M., D.J. O'CONNOR, R.V. THOMANN and J.L. MANCINI. 1975. Phytoplankton-zooplankton-nutrient interaction model for western Lake Erie. p.423-474. In B.C. Patten (ed.). *Systems Analysis and Simulation in Ecology*. Vol. III. Academic Press, N.Y.
- GILBERTSON, M., H.H. DOBSON and T.R. LEE. 1972. Phosphorus and hypolimnial dissolved oxygen in Lake Erie in Project Hypo. Appendix 3. CCIW Paper No. 6.

- HOHN, M.H. 1969. Qualitative and quantitative analyses of plankton diatoms, Bass Island area, Lake Erie, 1938-1965, including synoptic surveys of 1960-1963. Bull. Ohio Biol. Survey. New Series 3(1). 211 p. The Ohio State University, Columbus, Ohio.
- HÖRNSTRÖM, E., C. EKSTRÖM, U. MILLER and W. DICKSON. 1973. Försurningens inverkan på västkustsjöar. Information från Sötvattens-Laboratoriet, Drottningholm, Nr. 4, 81p.
- IMBODEN, D.M. 1974. Phosphorus model of lake eutrophication. Limnol. Oceanogr. 19:297-304.
- KARASIK, V. and I. STOVBUN. 1974. An apparatus for aerating ponds. Hydrobiol. J. 10:91-94.
- KIRCHNER, W.B. 1975. An examination of the relationship between drainage basin morphology and the export of phosphorus. Limnol. Oceanogr. 20:267-270.
- KIRCHNER, W.B. and P.J. DILLON. 1975. An empirical method of estimating the retention of phosphorus in lakes. Water Resour. Res. 11:182-183.
- LARSEN, D.P. and H.T. MERCIER. 1976. Phosphorus retention capacity of lakes. J. Fish. Res. Bd. Canada 33:1742-1750.
- MICHALSKI, M.F.P. and K.H. NICHOLLS. 1975. Phosphorus removal and water quality improvements in Gravenhurst Bay, Ontario. Ont. Min. Envir. Rep. 17p.
- MICHALSKI, M.F.P., K.H. NICHOLLS and M.G. JOHNSON. 1975. Phosphorus removal and water quality improvements in Gravenhurst Bay, Ontario. Verh. Internat. Verein. Limnol. 19:644-659.
- NICHOLLS, K.H. 1976a. Nutrient-phytoplankton relationships in the Holland Marsh, Ontario. Ecol. Monogr. 46:179-199.
- NICHOLLS, K.H. 1976b. Comparative limnology of Harp and Jerry Lakes, adjacent cottaged and uncottaged lakes on southern Ontario's Precambrian Shield. Ont. Min. Envir. Rep. 87p.
- NICHOLLS, K.H. 1977. The phytoplankton of the Kawartha Lakes. p.29-45+ 8 figs. In The Kawartha Lakes Water Management Study - Water Quality Assessment (1972-1976). Ont. Min. Envir. Rep. 185p.
- NICHOLLS, K.H. and P.J. DILLON. 1978. An evaluation of phosphorus-chlorophyll-phytoplankton relationships for lakes. Int. Rev. ges. Hydrobiol.: in press.
- NICHOLLS, K.H., D.W. STANDEN, G.J. HOPKINS and E.C. CARNEY. 1977. Declines in the near-shore phytoplankton of Lake Erie's western basin since 1971. J. Gt. Lakes Res., Internat. Assoc. Great Lakes Res. 3:72-78.

- O'CONNOR, D.J. and J.A. MUELLER. 1970. A water quality model of chlorides in Great Lakes. J. Sanit. Eng. Div. (ASCE) 96:955-975.
- RAINEY, R.H. 1967. Natural displacement of pollution from the Great Lakes. Science 155:1242-1243.
- SAKAMOTO, M. 1966. Primary production by phytoplankton community in some Japanese lakes and its dependence on lake depth. Arch. Hydrobiol. 62:1-23.
- SCHEIDER, W. and P.J. DILLON. 1976. Neutralization and fertilization of acidified lakes near Sudbury, Ontario. Water Poll. Res. Can. 11:93-100.
- SHAPIRO, J. 1973. Blue-green algae : why they become dominant. Science 179:382-384.
- SIRENKO, L., N. AVIL'TSEVA and V. CHERNOUSOVA. 1972. Effect of artificial aeration of pond water on the algal flora. Hydrobiol. J. 8:52-53.
- SNODGRASS, W.J. and C.R. O'MELIA. 1975. Predictive model for phosphorus in lakes. Env. Sci. Tech. 9:937-944.
- STRUS, R. 1976. Effects of artificial destratification on the zooplankton of Heart Lake, Ontario. Ont. Min. Envir. Rep. 18p.
- SWEERS, H.E. 1969. Removal of contaminants from Lake Ontario by natural processes. Proc. 12th Conf. Great Lakes Res.:734-741.
- VERDUIN, J. 1964. Changes in western Lake Erie during the period 1948-1962. Verh. Internat. Verein. Limnol. 15:639-644.
- VOLLENWEIDER, R.A. 1969. Möglichkeiten und Grenzen elementarer Modelle der Stoffbilanz von Seen. Arch. Hydrobiol. 65:1-36. (English translation by Fish. Mar. Serv. Canada).
- VOLLENWEIDER, R.A. 1975. Input-output models with special reference to the phosphorus loading concept in limnology. Schweiz. Z. Hydrol. 37:53-84.
- VOLLENWEIDER, R.A. 1976. Advances in defining critical loading levels for phosphorus in lake eutrophication. Mem. Ist. Ital. Idrobiol. 33:53-83.
- YAN, N.D., W.A. SCHEIDER and P.J. DILLON. 1977. Chemical and biological changes in Nelson Lake, Ontario, following experimental elevation of lake pH. Water Poll. Res. Can. 12:213-231.

APPENDIX A

A literature search has yielded 16 different P-chlorophyll relationships. The following is a condensed version of Nicholl's and Dillon's (1978) explanation of this variation, for which they concluded that there are two basic reasons:

1. Differences in laboratory and field techniques

The upper line in Fig. 31 is Sakamoto's original plot (1966) and his analyses are of total chlorophyll rather than just chlorophyll a. The four plots grouped in the middle used spring P levels while almost all others used mean summer or annual P concentrations. This difference clearly explains some of the variation in the relationships, particularly in view of the fact that Chapra and Tarapchak (1976) found that summer total phosphorus concentrations were consistently lower than spring values by a factor of 0.9 (Fig. 43), an observation that is well supported by Ontario lakes (Fig. 44).

In short, no two of the 16 studies used the same sampling periods, laboratory procedures and the same sampling zones in the lakes studied.

2. Chlorophyll a is no more than a crude approximation of algal biomass

It is now known that chlorophyll a content per unit of algal cell volume varies from 0.1 to 9.7%. Fig. 45 is a plot of results from a study of the Kawartha Lakes in southern Ontario (Nicholls 1977). It is obvious that switching from chlorophyll a concentration to volumetric algal biomass provides a stronger relationship. Similarly, Secchi disc biomass plots are better correlated than Secchi disc - chlorophyll a plots (Fig. 46). This superiority of 'measured' biomass is to be expected and is the better measurement of biomass even though a simple chemical measurement, like chlorophyll a or ATP, will still have a place in limnological studies because of their ease in execution.

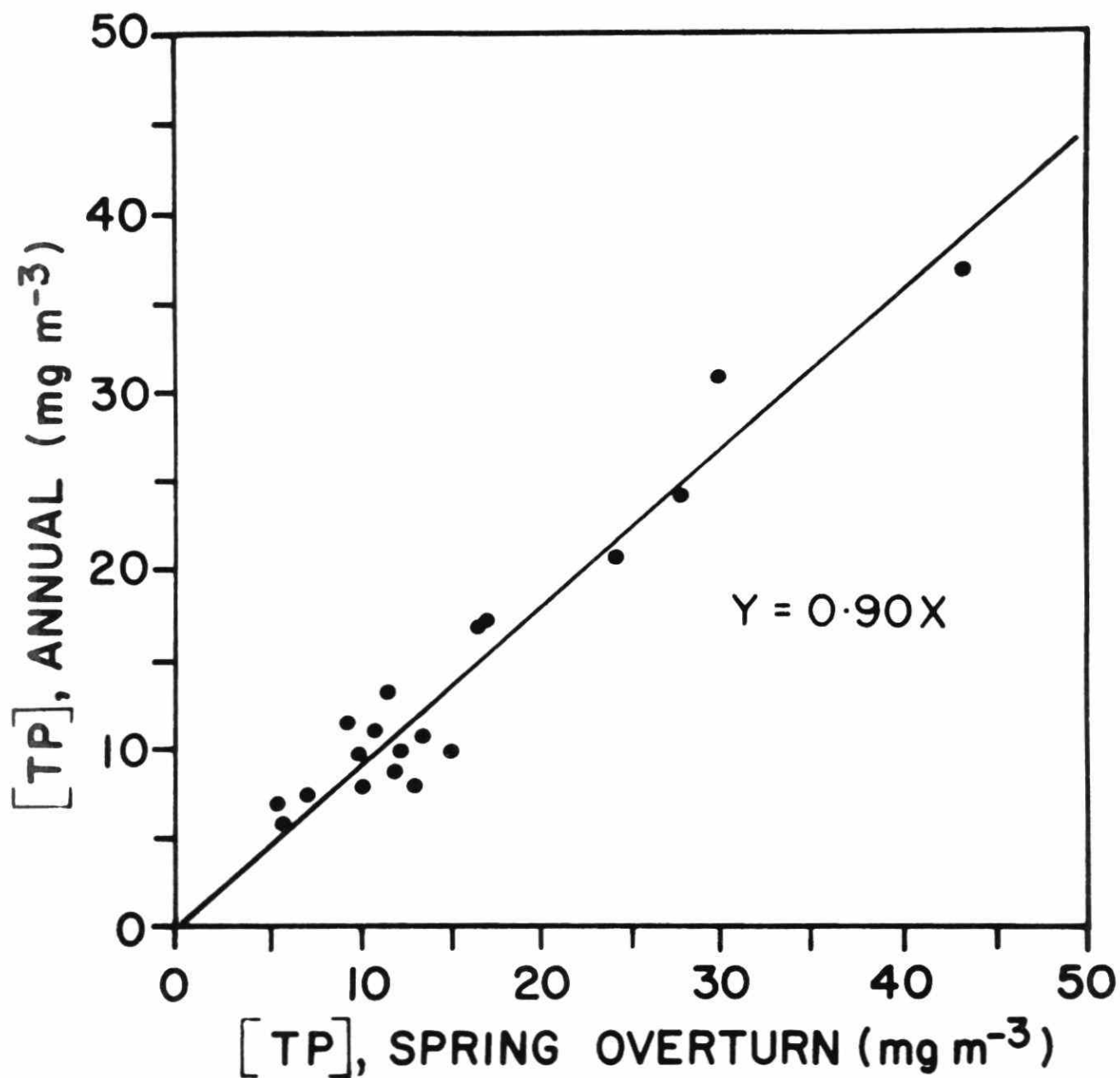


Fig. 43. Relationship between total phosphorus concentration measured at spring overturn and that calculated on an annual basis (from Chapra and Tarapchak 1976).

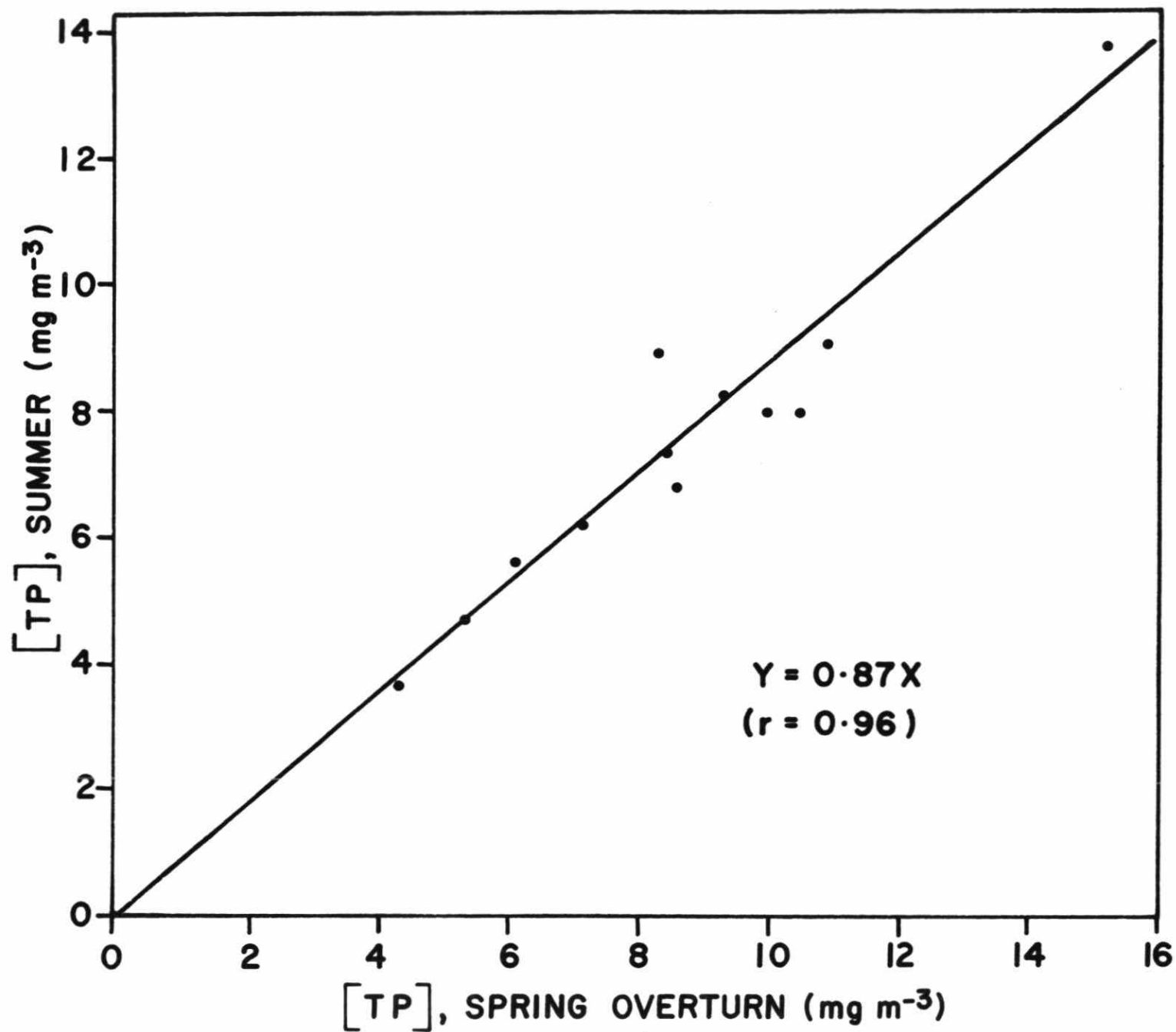


Fig. 44. Relationship between total phosphorus concentration measured at spring overturn and that calculated on an annual basis for lakes in southern Ontario.

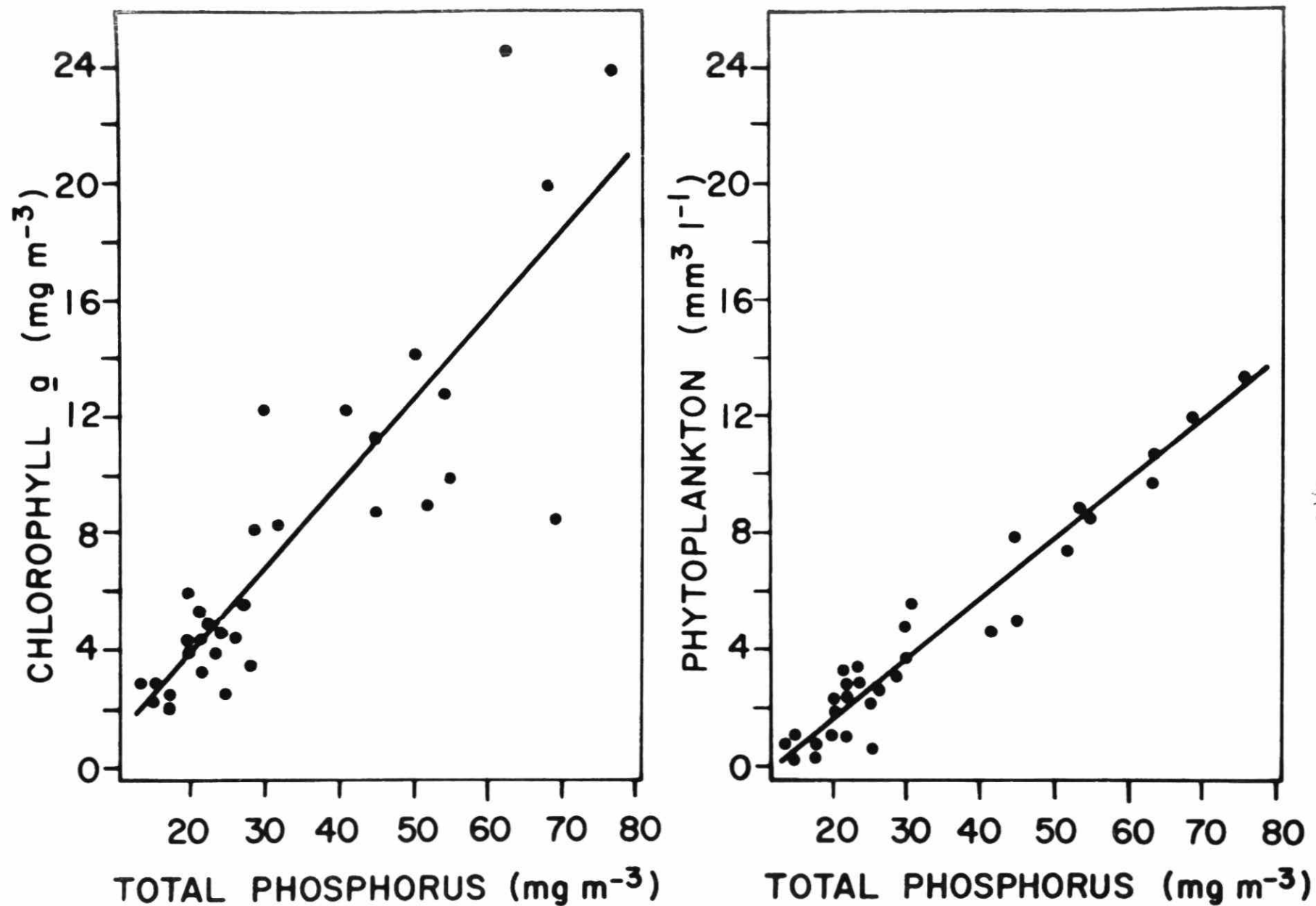


Fig. 45. Chlorophyll a concentration and phytoplankton biomass vs total phosphorus content for the Kawartha Lakes (Nicholls 1977).

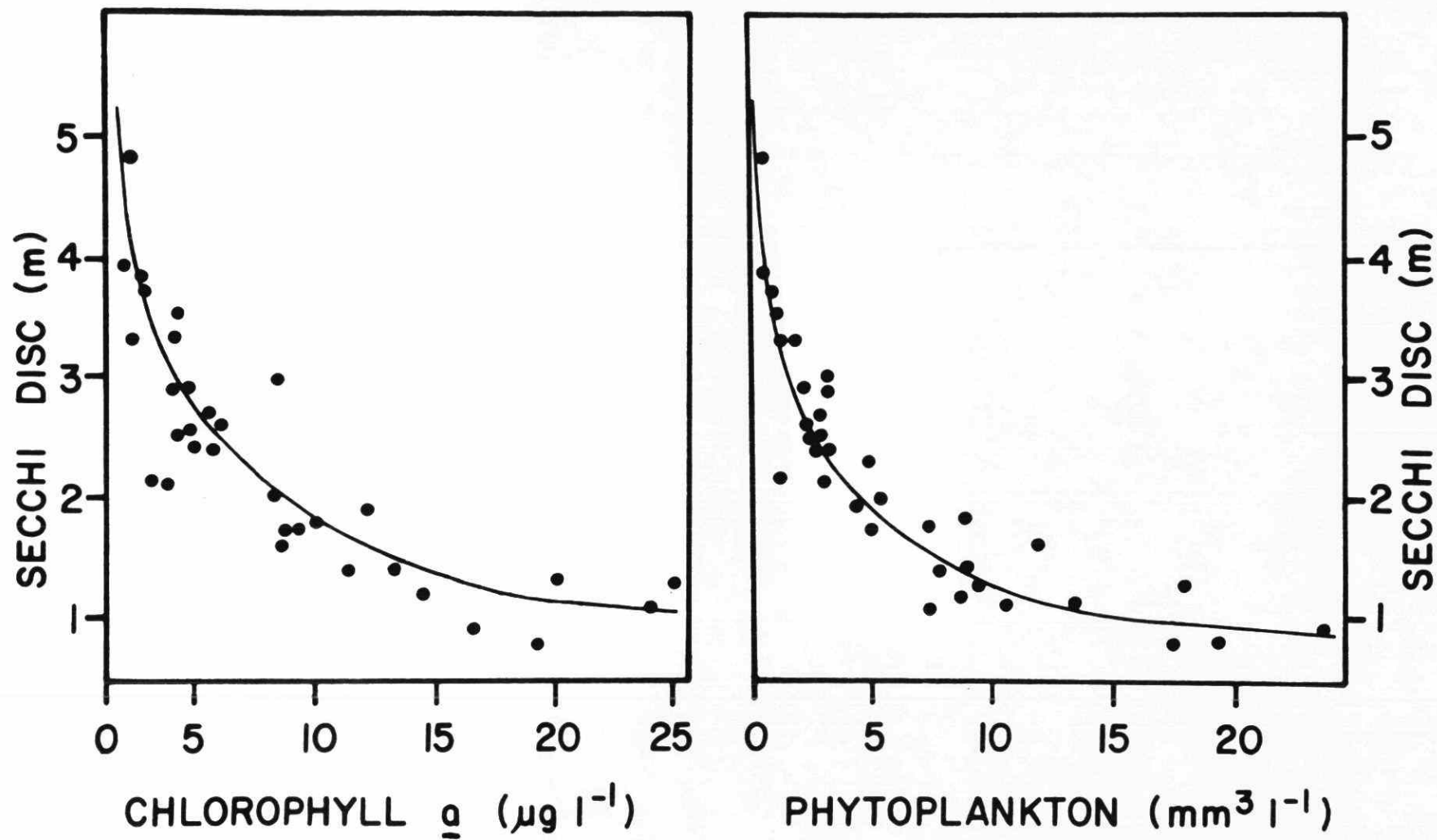


Fig. 46. Secchi disc depth vs chlorophyll a concentration and phytoplankton biomass for the Kawartha Lakes (Nicholls 1977).

